Ministry for Primary Industries
Manatū Ahu Matua

Aquatic Environment and Biodiversity Annual Review 2017

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

Growing and Protecting New Zealand

ACKNOWLEDGEMENTS

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The 2017 edition of the Aquatic Environment and Biodiversity Annual Review, the seventh in the series, updates previous editions but does not expand coverage. It summarises information on a range of issues related to the environmental effects of fishing and aspects of marine biodiversity and productivity relevant to fish and fisheries. This review is a conceptual analogue of the Ministry's annual Fisheries Assessment Plenary reports. It summarises the most recent data and analyses on particular aquatic environment issues and, where appropriate, assesses current status against any specified targets or limits. Whereas the Fisheries Assessment Plenary reports are organised by fishstock, the Aquatic Environment and Biodiversity Annual Review is organised by issue (e.g., protected species bycatch, benthic impacts), and almost all issues involve more than one fishstock or fishery.

Several Fisheries Assessment Working Groups (FAWGs) contribute to the Fisheries Assessment Plenary, but only three contribute substantially to the Aquatic Environment and Biodiversity Annual Review. These are the Aquatic Environment Working Group (AEWG), the Antarctic Working Group (ANTWG) and the Biodiversity Research Advisory Group (BRAG). A wide variety of research is summarised in the Aquatic Environment and Biodiversity Annual Review, and some of this is peer-reviewed through processes other than the Ministry's science working groups. In particular, the Department of Conservation funds and reviews research on protected species, and the Ministry of Business, Innovation and Employment and Universities fund a wide variety of research, some of which is relevant to fisheries. Where such research is relevant to fisheries and meets MPI's Research and Science Information Standard, it is considered for inclusion in the review.

Further expansion and improvement of this review is anticipated and additional chapters will be developed in the coming years to provide increasingly comprehensive coverage of the issues. The appendix summarising aquatic environment and marine biodiversity research since 1998 will be regularly updated. Data acquisition, modelling, and assessment techniques will also progressively improve, and it is expected that risk assessments and reference points to guide fisheries management decisions will be further developed. Both will lead to changes to the current chapters. We hope the condensation of information from numerous reports published on a variety of platforms into this annual review will continue to assist fisheries managers, stakeholders and other interested parties to understand the issues, locate relevant documents, track research progress and make informed decisions.

This revision has been led by the Science Group within the Directorate of Fisheries Management of the Ministry for Primary Industries (primarily Martin Cryer, Rich Ford, Barbara Graham, James Jolly, Mary Livingston, Jennifer Matthews, Ben Sharp, and Nathan Walker), but has also relied on the input of members of the AEWG, ANTWG and BRAG, as well as the Department of Conservation's Conservation Services Programme Technical Working Group and individuals who were commissioned to assist. I would especially like to recognise and thank the large number of research providers and scientists from research organisations, academia, the seafood industry, environmental NGOs, Māori customary, DOC and MPI, along with all other technical and non-technical participants in numerous AEWG, ANTWG, BRAG and CSP-TWG meetings for their substantial contributions to this review. My sincere thanks to each and all who have contributed.

I am pleased to endorse this document as representing the best available scientific information on the environmental effects of fishing and marine biodiversity, as at December 2017.

Martin Cryer

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CONTENTS

1 INTRODUCTION

1.1 CONTEXT AND PURPOSE

This document contains a summary of information and research on aquatic environment issues relevant to the management of New Zealand fisheries. It is designed to complement the Ministry's annual Reports from Fisheries Assessment Plenaries (e.g., the May plenary, MPI 2017a, and the November plenary, MPI 2017b) and emulate those documents' dual role in providing an authoritative summary of current understanding and an assessment of status relative to any overall targets and limits. However, whereas the Reports from Fisheries Assessment Plenaries have a focus on individual fishstocks, this report has a focus on aquatic environment fisheries management issues and biodiversity responsibilities that often cut across many fishstocks, fisheries, or activities, and sometimes across the responsibilities of multiple agencies.

This update has been developed by the Fisheries Science Team within the Marine Branch, Ministry for Primary Industries (MPI), with assistance from Working Group members and some research providers. As with the Reports from Fisheries Assessment Plenaries, it is expected to change and grow as new information becomes available and more issues are considered. This edition has no new chapters and, given other pressures this year, changes have been restricted to corrections and updates where new information is available.

The Annual Review has a broad, national focus on each issue and the general approach is to avoid too much detail at a local, fishery, or fishstock level. 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 that, although it might be locally important, might contribute only a small proportion to the total impact. The details of benthic impacts by individual fisheries are documented in selected chapters in the May or November Report from the Fisheries Assessment Plenary, and linked there to the fine detail and analysis in research reports.

The first part of this document describes the legislative and policy context for aquatic environment and biodiversity research commissioned by MPI, 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 five broad themes: protected species; non-QMS (mostly fish) bycatch; benthic effects; ecosystem issues (including New Zealand's oceanic setting); and marine biodiversity. A third part of the review includes appendices for reference. This review is not yet fully comprehensive in its coverage of all issues or of all research within each issue, but summarises the best available information on the issues covered. Each chapter has been reviewed by the appropriate working group at least once.

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. The main guidance to avoid, remedy, or mitigate any adverse effect of fishing on the aquatic environment is in sections 8, 9 and 15, although sections 10, 11 and 13 are also relevant to decision-making under this Act (Table 1.1). The Ministry also administers the residual parts of the Fisheries Act 1983, the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992, the Fisheries (Quota Operations Validation) Act 1997, the Maori Fisheries Act 2004, the Maori Commercial Aquaculture Claims Settlement Act 2004, the Aquaculture Reform (Repeals and Transitional Provisions) Act 2004, the Driftnet Prohibition Act 1991, and the Antarctic Marine Living Resources Act 1981. Other Acts are relevant in specific circumstances: the Wildlife Act 1953 and the Marine Mammals Protection Act 1978 for protected species; the Marine Reserves Act 1971 for 'no take' marine reserves; the Conservation Act 1987; the Hauraki Gulf Marine Park Act 2000; the Resource Management Act 1991 for issues in coastal marine areas that could affect fisheries interests or be the subject of sustainability measures under section 11 of the Fisheries Act; and the Exclusive Economic Zone and Continental Shelf Environmental Effects) Act 2012 for issues outside the Territorial Sea. These Acts are administered by other agencies and this leads to a requirement for MPI to work with other government departments (especially DOC, and through the Natural

Resource Sector[1]) and with various territorial authorities (especially Regional Councils) to a greater extent than is required for most fisheries stock assessment issues.

Table 1.1: Sections of the Fisheries Act 1996 relevant to the management of the effects of fishing on the aquatic environment and biodiversity.

Fisheries Act 1996
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, any 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 TACC 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.

[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.

Under the primary legislation lie various layers of Regulations and Orders in Council (see http://www.legislation.govt.nz). 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 1.2). 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.

Table 1.2: 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 recently been negotiated to facilitate management of non-highly migratory species in the South Pacific. • New Zealand Biodiversity Strategy 2000 and the Aichi Agreements. In 2016 New Zealand released an updated Biodiversity Action Plan which as goals and targets relevant to fisheries management.

1.3 POLICY SETTING

1.3.1 STRATEGIC INTENTIONS AND OUR STRATEGY

The Ministry for Primary Industries is the principal adviser to the government on agriculture, horticulture, aquaculture, fisheries, forestry, and food industries, animal welfare, and the protection of New Zealand's primary industries from biological risk. MPI's Strategic Intentions (formerly called Statement of Intent, SOI), document is an important guiding document for the short to medium term. That for 2015–20 is available on the Ministry's website at: http://www.mpi.govt.nz/document-vault/9602.

Aspects of the Strategic Intentions document (SI) of key relevance to fisheries include supporting the understanding of sustainable limits to natural resource use as part of Medium-Term Outcome 5 *The primary sector maximises the use and productivity of natural resources within environmentally sustainable limits and is resilient to adverse climatic and biosecurity events*. The primary industries are the largest user of natural resources, and MPI is an important regulator and adviser on their sustainable use. How we all use and manage these natural resources affects New Zealand's future prosperity and the natural capital that underpins New Zealand's production systems. The SI notes a growing demand for more support for the primary sector's market claims about product attributes that help capture premium prices, including assurances about sustainability in fishing. For these claims to be effective, they need to be underpinned by science-based, transparent, verifiable information.

Another important role is supporting third-party certification of fisheries by, for example, the Marine Stewardship Council as part of Medium-Term Outcome 1 *Export success is enhanced by the integrity of primary sector products and increasing the use of New Zealand's unique culture and brand*. New Zealand's export sectors derive significant benefits (including lower market access costs) and competitive advantage from New Zealand's reputation for safe and suitable food, favourable animal and plant health status and market assurances. To leverage these advantages, MPI needs new ways of assisting New Zealand exporters to access and succeed in international markets and gain additional export value from the New Zealand brand, including its Māori dimension.

The Ministry's broad approach was updated in 2017 with a refresh of *Our Strategy 2030*. The new strategy was called *Our Strategy* (Figure) 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, MPI contracts the following types of research (relevant to this document):

- Aquatic environment research to assess the effects of fishing on marine habitats, protected species, non-target species of fish, and to understand habitats of special significance for fisheries;
- Marine biodiversity and productivity research to increase our understanding of the systems that support resilient ecosystems and productive fisheries, including their trophic linkages and the effects of climate change.

Figure 1.1: MPI's *Our Strategy*, released in 2017

1.3.2 FISHERIES PLANS

Fisheries planning processes for deepwater, 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 Highly Migratory Species were approved by the Minister in September 2010 and a suite of three draft plans for inshore species was released in July 2011. MPI 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 Ministry's websites 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.).

1.3.3 OTHER STRATEGIC DOCUMENTS

A number of strategies or reviews have been published that potentially affect fisheries values and research. These include: the New Zealand Biodiversity Strategy (2000); the Biosecurity Strategy (2003, followed by its science strategy 2007); the MPA 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 by MPI in 2013); and the New Zealand National Plan of Action for the Conservation and Management of Sharks (2013); New Zealand's Biodiversity Action Plan 2016. Links to these documents are provided in Appendix 19.8 because they provide some of the broad policy setting for aquatic environment issues and research across multiple organisations and agencies.

In 2012, the Natural Resource Sector cluster formed a Marine Director's Group to improve data sharing and information exchange across key agencies with marine environmental responsibilities, particularly MPI, DOC, MfE, EPA, LINZ, MBIE and Stats NZ.

1.4 SCIENCE PROCESSES

1.4.1 RESEARCH PLANNING

The Ministry has adopted an overall approach of specifying 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 (on protected species issues), MPI 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. Marine biodiversity and productivity research has a much broader and more strategic focus and planning of such research is conducted with assistance from the Biodiversity Research Advisory Group (BRAG).

1.4.2 RESEARCH REVIEW AND CONTRIBUTING WORKING GROUPS

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 MPI'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. MPI's Fishery Assessment Working Groups occasionally consider research relevant to this review where there is particular relevance to a fishery. Terms of Reference for MPI working groups are periodically revised and updated (see Appendices 19.1–19.6 for those of working groups relevant to this document).

AEWG is convened for MPI's 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 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; fishing-related changes to ecosystem structure and function, including trophic effects; and effects of aquaculture and fishery enhancement on the environment and on fishing. Where possible, 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 AEWG is open (attendees for 2017 are listed in Appendix 19.1).

The ANTWG is convened for the Ministry's peer review purposes with an overall purpose of assessing, based on scientific information, the stock status and the effects of fishing for Antarctic fisheries. The purview of 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 fishing-related changes to ecosystem structure and function, 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 (attendees for 2017 are listed in Appendix 19.3).

The two main responsibilities of BRAG are: to review, discuss, and convey views on the results of marine biodiversity research projects contracted by MPI; and to discuss, evaluate, make recommendations and convey views on Medium Term Biodiversity Research Plans and constituent individual projects. Both tasks have hitherto been undertaken in the context the strategic goals in the New Zealand Biodiversity Strategy (2000) and the Strategy for New Zealand Science in Antarctica and the Southern Ocean (2010), but the focus of the programme has recently been aligned with more recent strategic documents. BRAG also provides advice and oversight of cross-government projects such as NORFANZ, BIOROSS, Fisheries and Biodiversity Ocean Survey 20/20; and International Polar Year (IPY) Census of Antarctic Marine Life (IPY-CAML).

Following consideration at one or more meetings of appropriate working groups, reports from individual projects are also technically reviewed by the Ministry 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 (Fisheries Assessment Research Documents) and MBBRs (Marine Biodiversity and Biosecurity Reports), and some FRRs can be found 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 MPI

Almost all research of direct relevance to management of fish stocks is commissioned by MPI and reviewed through MPI fishery assessment working groups. This is a structured approach to meeting the requirements of the RSIS. However, research on various aspects of the environmental effects of fishing is commissioned by a much wider range of 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. MPI working groups, including AEWG, 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. 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 Ministry's fisheries science team before they can be used.

1.5 REFERENCES

Ministry for Primary Industries (2017a) Fisheries Assessment Plenary, May 2017: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 1556 p.

Ministry for Primary Industries (2017b) Fisheries Assessment Plenary, November 2017: stock assessments and stock status. Compiled by the Fisheries Science Group, Ministry for Primary Industries, Wellington, New Zealand. 500 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

The Ministry has identified five broad categories of research on the environmental effects of fishing (Figure 2.1): incidental capture and fishing-related mortality of protected species; bycatch of non-protected species, primarily non-QMS fish; modification of benthic habitats (including seamounts); and various ecosystem effects (including fishing and non-fishing effects on habitats of particular significance for fisheries management, effects of aquaculture on the environment and wild-capture fisheries, trophic relationships and Antarctic Science). A risk assessment chapter has also been added prior to the research themes, due to its cross-cutting nature, although this structure may be reconsidered in future. Other emerging issues (such as the genetic consequences of selective fishing) are not dealt with in detail in this edition but it is anticipated that those that turn out to be important will be dealt with in future iterations. The fifth theme for this document is MPI research on marine biodiversity. The research has been driven largely by the Biodiversity Strategy but has strategic importance for fisheries in that it provides for better understanding of the ecosystems that support fisheries productivity.

Our understanding is not uniform across these themes and, for example, our knowledge of the quantum and consequences of fishing-related mortality of protected species is much better developed than our knowledge of the consequences of mortalities of non-target fish, bottom trawl impacts, or land management choices for ecosystem processes or fisheries productivity. Ultimately, the goal of research described in this synopsis is to complement information on fishstocks to ensure that the Ministry has the information required to underpin the ecosystem approach to fisheries management envisaged in the future. 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 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. The Ministry explored better ways to document, review, publicise, and integrate information from environmental assessments with traditional fishery assessments, including annual publication of this document. This will rely 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, also on results published by other organisations and in the scientific literature. 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	RESEARCH QUESTIONS	CURRENT WORK
1.PROTECTED SPECIES • Marine mammals • Seabirds • Turtles • Protected fish • Corals	• How many of each NZ-breeding protected species are caught and killed in our fisheries (and out of zone)? • How many unobserved deaths are caused? • What is the likely effect of fishing-related mortality on protected species populations? • Which species/populations are most at risk? • Which fisheries cause the most risk and where are the most cost-effective gains in mitigation to be made? • What mitigation approaches are most successful and in what circumstances? • What levels of bycatch would lead to different population outcomes?	• Estimation of annual bycatch of protected species by fishery • Abundance and productivity of key seabird populations • Abundance and productivity of Hector's & Māui dolphins • Semi-quantitative risk assessment for all seabirds • Semi-quantitative risk assessment for marine mammals • Full quantitative risk assessment for selected populations • Modelling to assess links between bycatch and population outcomes
2. OTHER BYCATCH • Non-QMS fish & invertebrates	• How much non-target fish is caught and discarded in our fisheries? • What is the effect of that bycatch? • What do trends in bycatch show?	• Continued monitoring cycle for deepwater and highly migratory • Data collection to allow improved risk assessment for sharks and Non-QMS fish
3. BENTHIC EFFECTS • Distribution of habitats & trawling • Effects of trawling on each	• What seabed habitats occur where in our TS/EEZ and how much of each is affected by trawling or shellfish dredging? • How sensitive is each habitat to disturbance and how do we lose (or gain) when each is disturbed? • What are the likely consequences of different management approaches? • What are benthic recovery times in different habitats?	• Testing habitat suitability models at a range of spatial scales • Monitoring and recovery rate of some key inshore habitats • Assessment of relative sensitivity of habitats • Mapping of deepwater and inshore trawl footprints • Development of a risk assessment
4. ECOSYSTEM EFFECTS • Trophic studies • Habitats of significance • Ecosystem indicators • Land-use effects • Climate variability • Climate Change • System productivity	• How do the ecosystems that support our fisheries function? • What are the key predator-prey or synergistic relationships in these systems? • Are our fisheries affecting food webs or ecosystem services? • What changes are occurring in the ecosystems that support our fisheries? • What is "habitat of particular significance for fisheries management"? • How do fisheries and/or land management affect fish habitat and fisheries production?	• Habitat of significance: Marlborough Sounds blue cod habitat • Monitoring and indicators of environmental change for deepwater fisheries • Much of the more strategic work in this theme is conducted under the biodiversity programme (see below)
5. MARINE BIODIVERSITY • Characterising NZ biodiversity • Functional ecology • Genetic diversity • Ocean climate • Metrics & indicators • Threats & impacts • Ross Sea	• What are the key drivers of pattern in New Zealand's marine biodiversity? • How does biodiversity contribute to the resilience of ecosystems to perturbation and climate change? • What are the major risks and opportunities from ocean-climate variability and trends? • What drives genetic connectivity within species? • What do we need to measure and monitor to assess risks and change? • How are biota adapted to polar conditions and what is their sensitivity to perturbation?	• Climate change risk & opportunity for the seafood industry • SPRFMO benthic habitats • Genetic connectivity between New Zealand and the Louisville Ridge • Ocean acidification modelling • Experimental response of shellfish to warming and acidification • Monitoring surface plankton • BPA biodiversity • Sublethal effects of OA on fish productivity • Coralline and macroalgae habitats

Figure 2.1: Summary of themes in the Aquatic Environment and Biodiversity Annual Review 2016. [Continued on next page]

CURRENT STATE OF KNOWLEDGE

- Aggregate "on deck" captures of seabirds (and approximate species composition), marine mammals, and large sharks known reasonably well for offshore trawl and longline fisheries, but less well for inshore fisheries (where observer coverage has historically been low).
- Incidental, cryptic, or unobserved mortality are poorly known (and difficult to assess).
- Factors affecting fishing related mortality are well known for most seabirds and marine mammals.
- Knowledge of population abundance is increasing for some key seabird species and well known for sea lions, but poorly known or dated for other seabirds, some species of dolphins, fur seals and most sharks.
- Semi-quantitative risk assessments on all seabird and marine mammal species, fully quantitative risk assessments have been completed for five seabird species, Hector's / Maui dolphins and sea lions.
- Impact of fishing-related mortality on most protected species remains uncertain because of some key knowledge gaps.
- Some methods of mitigating fishing-related mortality have been formally tested.

- Bycatch and discards are monitored and reported using observer records for the main deepwater and highly migratory fisheries.
- Bycatch and discards for inshore vessels remain poorly known.
- Some mitigation approaches have been assessed (e.g., for scampi trawl).

- Modelled predictions are available of the distribution of seabed habitats at a broad scale using classifications (BOMEC) and at finer scale for seamounts and some biogenic habitats.
- Excellent understanding of the distribution of bottom trawling in offshore waters, good understanding in coastal waters, but poor understanding for most shellfish dredge fisheries.
- Good understanding of the effects of trawling on some nearshore and deepwater habitats.
- General understanding of the effects of trawling on biogeochemical processes.
- General understanding of the relative sensitivity of different habitats.
- Tools to design candidate spatial management measures have been developed and tested (including for High Seas fisheries)

- The nature and variability in the diets of key commercial species in the Chatham Rise ecosystem are reasonably well understood.
- A preliminary trophic model of the Sub-Antarctic ecosystem suggests a low productivity system supporting a simple food chain with high transfer efficiencies.
- The distribution of spawning, pupping, egg-laying, and juveniles of key species has been described in public atlases.
- Land-based effects on fish habitat and coastal biodiversity has been reviewed and documented.
- Basic monitoring of ecosystem change over time (through fish-based indicators calculated from trawl survey data and acoustic time series of mesopelagic biomass) has started.
- Ocean climate variability and historical change are reasonably well understood.
- Broad reviews have been completed of the impacts of climate variability on fisheries (especially recruitment), but the likely impacts of ocean climate change or acidification remain poorly known.
- This theme has links and synergies with MBIE, DOC, universities and the MPI biodiversity programme.

- Taxonomy and ID Guides have been produced and specimens recorded in National Collections.
- Biodiversity surveys completed on local scale (Fiordland, Spirit's Bay, seamounts) and larger fishery scale (Norfolk Ridge, Chatham Rise, Challenger Plateau, Bay of Islands).
- Measures and indicators for marine biodiversity measures and ecosystem have been developed.
- Predictive modelling techniques have been applied and habitat classification methods improved
- Productivity in benthic communities has been measured.
- Specimens from New Zealand have been genetically assessed and entered into the barcode of life.
- Seamount connectivity, land-sea connectivity, and endemism have been studied.
- Two measures of marine conditions and marine biodiversity have been evaluated as tier 1 statistics.
- Demersal fish trophic studies on the Chatham Rise have been completed.
- A review of NZ data from deep-sea and abyssal habitats has been completed.
- A multidisciplinary study of long-term (1000 years) changes to NZ marine ecosystem is complete.
- Latitudinal gradient project, ICECUBE and 2 large scale surveys in the Ross Sea have been conducted.
- The MPI Marine Biodiversity Research Programme theme has links and synergies with MBIE, DOC, universities and the MPI AEWG programmes










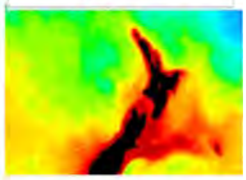

Figure 2.1 [Continued]: Summary of themes in the Aquatic Environment and Biodiversity Annual Review 2016.

3 SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT (SEFRA):

A FRAMEWORK FOR QUANTIFYING AND MANAGING INCIDENTAL COMMERCIAL FISHERIES IMPACTS ON NON-TARGET SPECIES

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.

Note: This chapter is reprinted with only slight amendments from its original publication in the AEBAR 2016. Following incorporation of suggestions made by a panel of independent international reviewers in 2017 (Lonergan et al. 2017) this paper will be submitted for external publication. Lead author for correspondence: ben.sharp@mpi.govt.nz

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 generally low, and what data are available are 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 seabirds 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 is 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 trouble spots and target management interventions effectively, to track location- or fishery-specific change over time, and to equitably assign responsibility for necessary changes to fishing practices. 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 further research). 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 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).

Where further structural or methodological improvements to the SEFRA method have been identified by MPI scientists, these are included in the method description below. Some of these changes have already been discussed in the Aquatic Environment Working Group and appear in the 2017 iteration of the NZSRA (Richard et al. 2017); other changes have yet to be incorporated and are described below to guide future work.

In the National Plan of Action – seabirds (NPOA-Seabirds; Ministry for Primary Industries 2013), the SEFRA method was formally adopted as the means by which species-level risk to seabirds is assessed, and the 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:

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

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:

$$D_{sg} = C_{sg} * \varkappa_{sg} \quad (11)$$

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

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.,

$$I_{zg} = C_{zg} * k_{zg} = q_{zg}\Theta_{zg}k_{zg} \quad (12)$$

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).

$$v_{zg} = q_{zg}k_{zg} \quad (13)$$

$$I_{zg} = v_{zg}\Theta_{zg} \quad (14)$$

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:

$$I_{zg} = v_z v_g \Theta_{zg} \quad (15)$$

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:

$$C_{zg} = \frac{v_z v_g \Theta_{zg}}{k_{zg}} \tag{16}$$

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:

$$C_{zg} \sim Poisson(\hat{\lambda}_{zg}) \tag{17}$$

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),

$$\widehat{\lambda'}_{zg} = \sum_{i} \frac{v_z v_g \Theta'_{zgi}}{k_{zg}} * \epsilon_{zg} \tag{18}$$

where the ' 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:

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

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:

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

(23)

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

(24)

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

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

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

(25)

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

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

(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.,

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

(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:

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

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 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 $N_{\mathbf{min}}$ 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 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 eliminates the use of the recovery factor *f*. The previous incorporation of *f* into PBR in equation (29) effectively confused risk assessment and risk management within the estimation of 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 improved 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 setting management goals is more appropriately addressed in the choice of population recovery outcome inherent in the definition of φ in equation (30); defining different management objectives for different species is best addressed outside the risk assessment. 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 defined with explicit reference to a population recovery or stabilisation outcome, and is estimated by simulation, including consideration of uncertainty and the effects of environmental stochasticity. This approach was first developed by Richard & Abraham (2013a). For the NZSRA and NZMMRA, New Zealand has defined a population reference outcome as follows: 'for impacts equal to or lower than PST (i.e., $R \leq 1$), the population will recover to or stabilise at a level at or above 50% of carrying capacity, with 95% certainty (and considering the effects of environmental stochasticity)'.

Alternative population recovery outcomes are of course possible, to reflect for example different management goals or different levels of tolerance for uncertainty; these would be expected to vary especially between risk assessments devoted to different kinds of taxa (e.g., for non-target fish vs. for marine mammals). Corresponding values of φ can be explored using population simulations particular to the taxa in question.

The original seabird-focused simulations in Richard & Abraham (2013a) used this same population recovery outcome above corresponding to R=1, but with a different mathematical formulation (with an adjustment parameter called ρ), in which the ρ parameter performed more than one function simultaneously, i.e., correcting biological parameter estimation bias as well as adjusting for the population recovery outcome. The improved PST formulation in the NZMMRA and updated NZSRA is designed to separate these roles so that φ serves only as an adjustment factor reflecting the population outcome; the parameter bias correction function also previously addressed within ρ is now addressed separately via improved parameter estimation methods first reviewed by the AEWG in 2016 (see Chapter 8).

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.

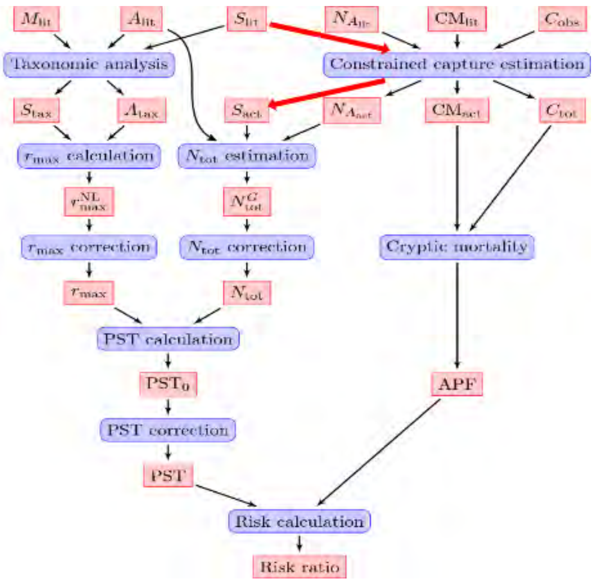

Figure 3.1: Diagram of data inputs and calculation pathways utilised in the current (2017) iteration of the NZSRA. The use of (realised) adult survival estimates to constrain captures and fishery-related deaths is highlighted.

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 mackeral 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

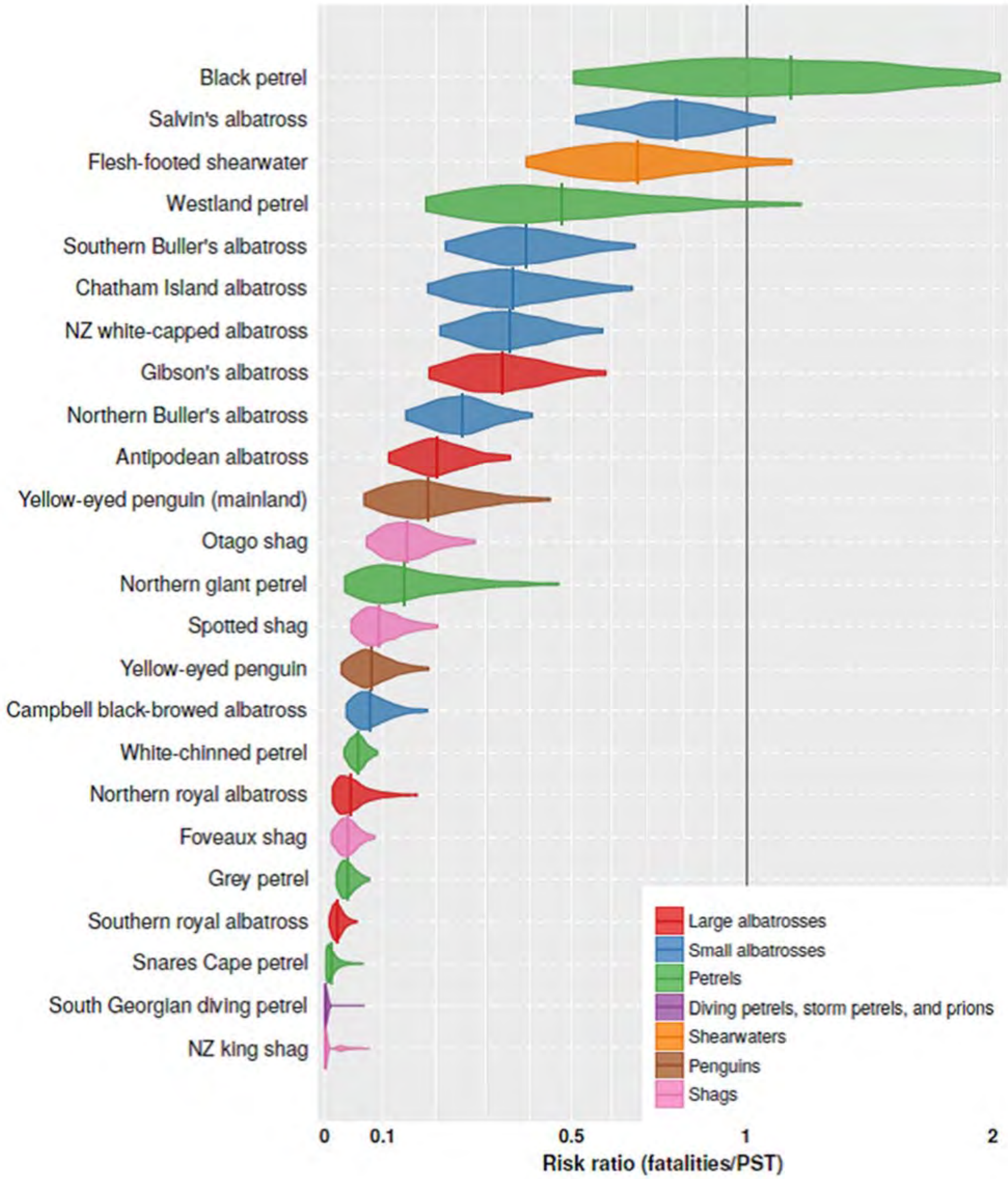

Figure 3.2: Standard species-level output of the NZSRA (from Richard et al. 2017). Species risk is shown on the x axis; the vertical line at R=1 corresponds to the level of *all human-induced mortality* that the species can sustain while still meeting the population recovery or stabilisation outcome inherent in the definition of the PST. Grey distributions illustrate change since the previous risk estimates.

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.

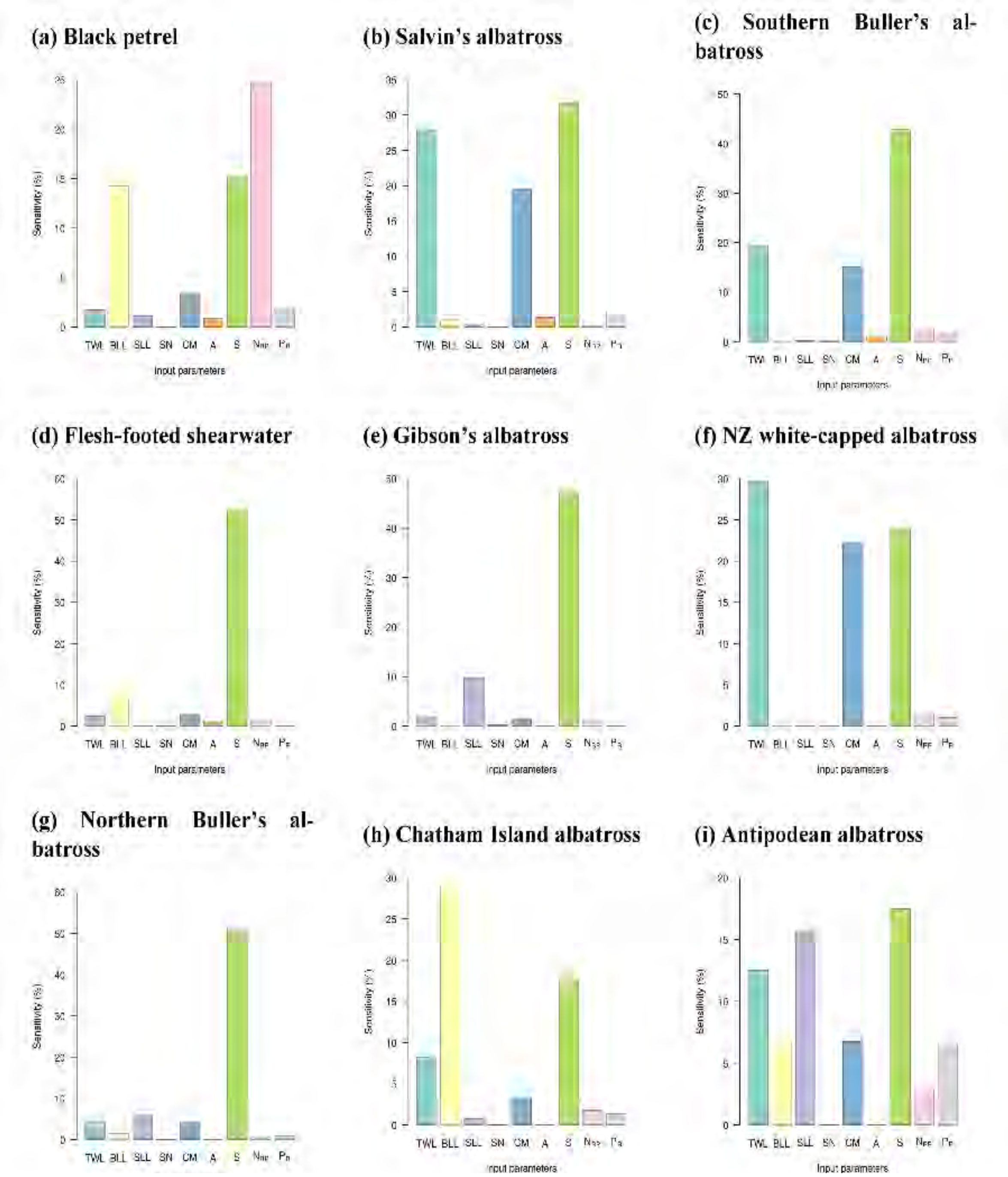

Figure 3.3: Example plot showing the propagation of uncertainty from uncertain input parameters through to output estimates of risk for at-risk species in the NZSRA. Parameters contributing to high levels of output uncertainty become a priority for future research. Note that this figure derives from 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., Figure 3.6 and Figure 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). 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).

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

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

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.

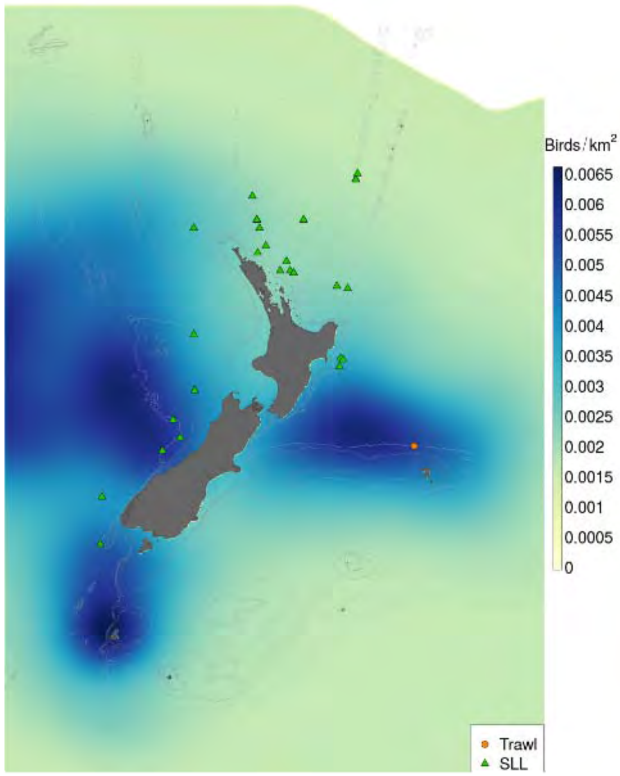

Figure 3.4: Spatial distribution layer (derived from global tracking data) for Gibson's albatross. Capture events are also shown.

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 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 (Figure 3.6 and Figure 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 Start	Breeding period End	Biological N	% staying in NZ
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 3.5. The intersection of the species and effort distributions (Figure 3.4 and Figure 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

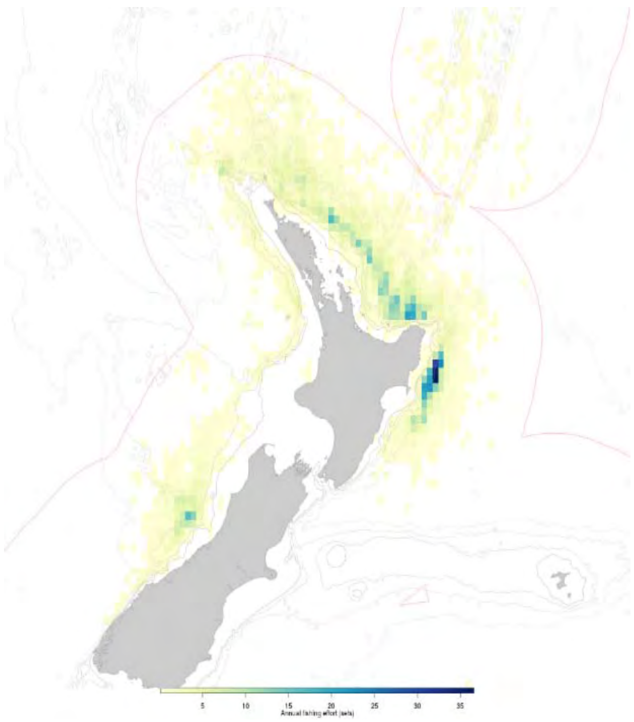

Figure 3.5: Fishing effort spatial distribution for the small (domestic) SLL fishery group not targeting swordfish, 2005–06 to 2014–15.

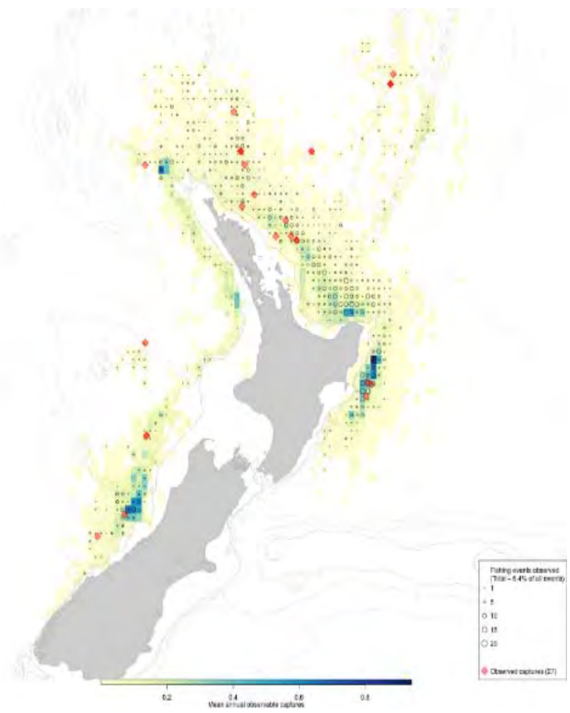

Figure 3.6: Overlap *O* of the distribution of Gibson's albatross (Figure 3.4) with the small (domestic) SLL fishery group (Figure 3.5). Black circles denote observed fishing effort. Capture events are also shown. Captures are expected to occur in space proportional to the intensity of the overlap in 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).

number of hauls only, independent of distance or duration (suggesting that most protected species captures occur at 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 c*ryptic 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.

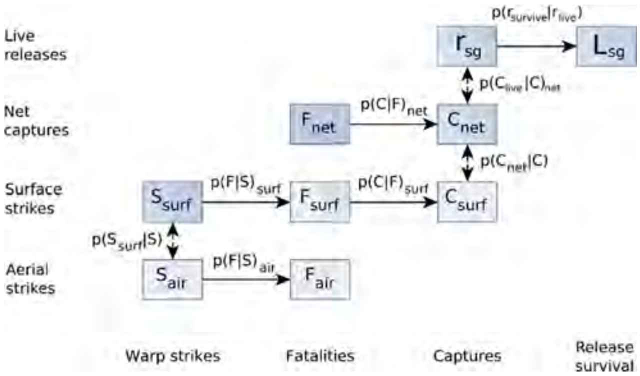

Figure 3.7: Transition probabilities by which the total fisheries-related deaths multiplier $\varkappa_{sg}$ (including live release survival as in equation (10)) is estimated for seabirds in trawl fishery groups.

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* 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 $k_{sg} - set$.

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 Table 3.3 and Table 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.

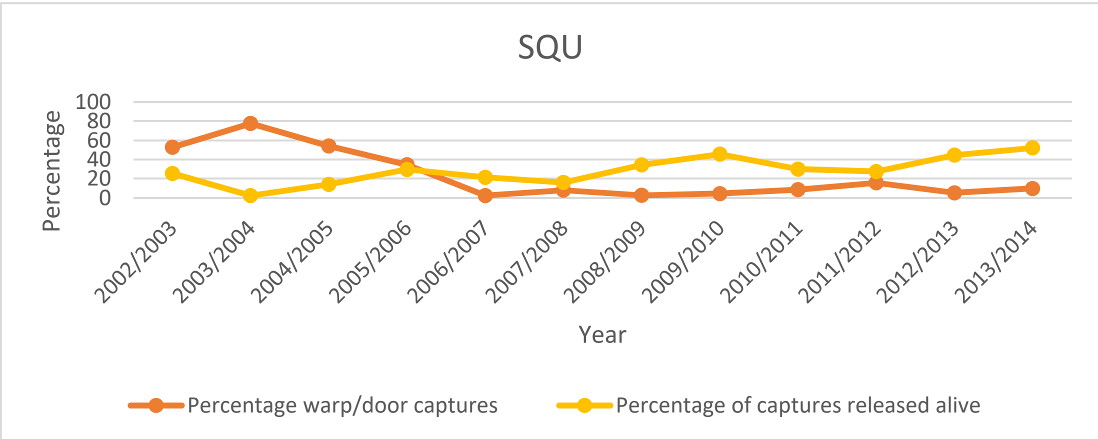

Figure 3.8: Declining proportion of seabird captures on the warp vs. on the net, and increasing proportion of seabirds released alive, in the well-observed squid fishery, corresponding to the implementation of changed practices from approximately 2003–05. Changes of this kind may result in significant 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 Mean	Prior 95% c.i.	Posterior Mean	Posterior 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

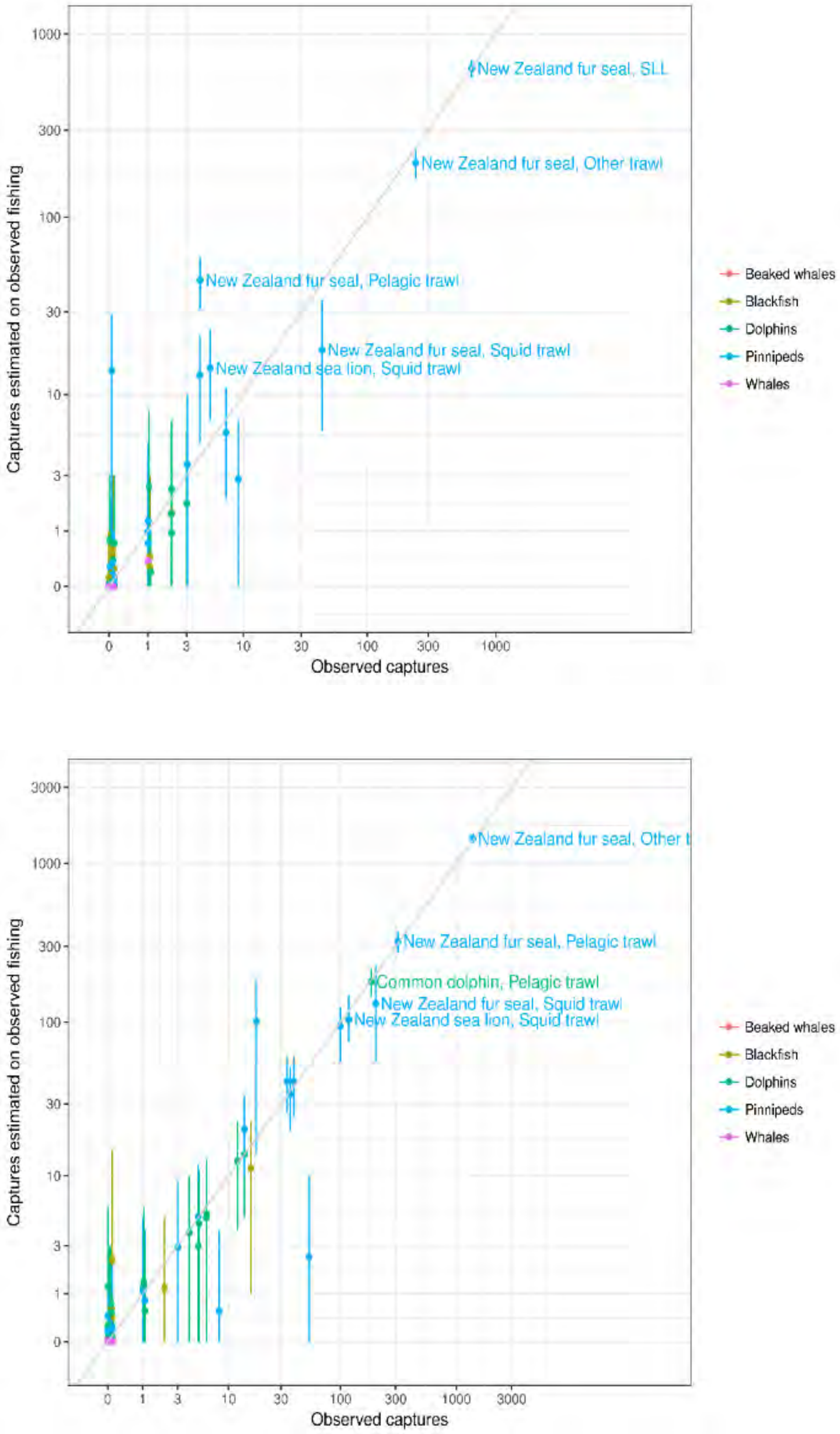

Figure 3.9: Example model diagnostics plot showing observed vs. expected numbers of live captures (top) and dead captures (bottom) for each fishery and species group combination in the 2017 NZMMRA.

3.4 ALTERNATIVE IMPLEMENTATIONS OF THE SEFRA FRAMEWORK

Alternative 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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THEME 1: PROTECTED SPECIES

4 NEW ZEALAND SEA LION (*PHOCARCTOS HOOKERI*)

Scope of chapter	This chapter briefly describes: the biology of New Zealand sea lion/rāpoka (*Phocarctos hookeri*) 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	Southern parts of the New Zealand EEZ and Territorial Sea, near the Subantarctic Islands and around Stewart Island and South Island.
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	Understanding 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; improved estimation of cryptic mortality in trawls employing SLEDs.
Emerging issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries potentially interacting with Stewart Island and mainland colonies.
MPI research (current)	PRO2017-10 *Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries;* PRO2017-08C *Factors affecting New Zealand sea lion pup survival*; SEA2016-24 *Supplemental sea lion population modelling to support an updated Squid Trawl Fishery Operational Plan*; SEA2015-12 *Potential impacts of fisheries restrictions for the New Zealand sea lion TMP*; SEA2015-10 *Sea lion prey survey*; PRO2013-01 *Estimating the nature & extent of incidental captures of seabirds, marine mammals & turtles in New Zealand commercial fisheries*; SEA2014-12 *New Zealand sea lion stable isotope analysis*; PRO2012-02 *Assessment of the risk to marine mammal populations from New Zealand commercial fisheries*; PRO2014-02 *Quantitative risk assessment for New Zealand sea lions*; *SEA2014-12 New Zealand sea lion stable isotope analysis.* Joint funding with Deepwater Group; *New Zealand sea lion population project (Campbell Islands).*
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2014-01 *Observing commercial fisheries*; INT2015-02 *Identification of marine mammals, turtles and protected fish captured in New Zealand fisheries*; MIT2014-01 *Protected species bycatch newsletter.* POP2015-05 *New Zealand Sea Lion: Auckland Islands population project.* NIWA Research: SA123098 *Multispecies modelling to evaluate the potential drivers of decline in New Zealand sea lions*; TMMA103 *Conservation of New Zealand's threatened iconic marine megafauna.*
Related chapters/issues	Chapter 5: New Zealand fur seals.

Note: This chapter has been updated for the 2017 AEBAR with recent population studies, capture estimation and risk assessment results.

4.1 CONTEXT

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. Under s.3E of the MMPA or s.14F of the Wildlife Act 1953, the Minister of Conservation, with the concurrence of the Minister of Fisheries, may approve a population management plan (PMP). Although a New Zealand sea lion PMP was proposed by the Department of Conservation (DOC) in 2007 (DOC 2007), following consultation DOC decided not to proceed with the PMP.

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. New Zealand's subantarctic islands, including Auckland and Campbell islands, were inscribed as a World Heritage area in 1998.

The Minister of Conservation gazetted the New Zealand sea lion as a threatened species in 1997.

In the absence of a PMP, the Ministry for Primary Industries (MPI) manages fishing-related mortality of New Zealand sea lions under s.15 (2) of the FA. Under that section, the Minister '*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 (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand sea lions is Management Objective 2.5: *Manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened and protected species*.

Specific objectives for the management of incidental captures of New Zealand sea lions are 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 (Ministry of Fisheries 2010). These fisheries include trawl fisheries for arrow squid (SQU 1T and SQU 6T), southern blue whiting (SBW) and scampi (SCI). The SBW chapter of the National Deepwater Plan is complete and includes Operational Objective 2.2: *Ensure that incidental New Zealand sea lion mortalities, in the southern blue whiting fishery at Campbell Island (SBW6I), do not impact the long term viability of the sea lion population and captures are minimised through good operational practices*. The chapters in the National Deepwater Plan for arrow squid and scampi are under development.

The New Zealand sea lion population is monitored based on pup counts at the main breeding colonies, the largest of which are on 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 TMP was finalised in 2017 (DOC & MPI 2017). It was a culmination of 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, reflecting the demographic population model and multi-threat risk assessment described in Roberts & Doonan (2016), recognises that no single identified threat in isolation is responsible for the observed population decline, such that recovery will benefit from mitigation of multiple threats at the four main breeding sites (DOC & MPI 2017). The TMP commits to two objectives:

1) Halt the decline of the New Zealand sea lion population within 5 years.
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 proposes 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.

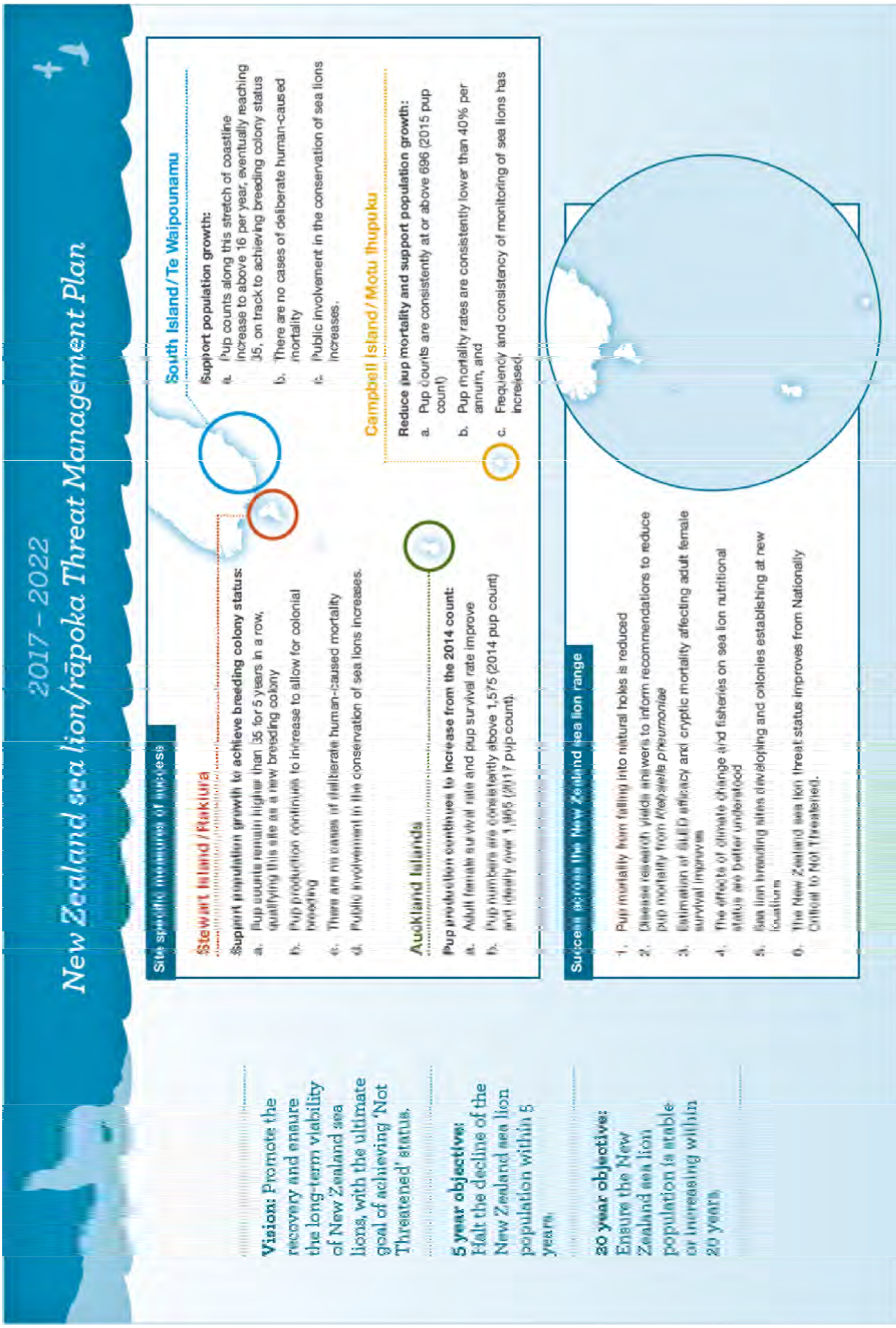

Figure 4.1: Threat management and population recovery objectives specific to four different New Zealand sea lion breeding populations, from the New Zealand sea lion Threat Management Plan (DOC & MPI 2017).

Figure 4.2: Workstreams identified in the New Zealand sea lion Threat Management Plan (DOC & MPI 2017).

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.

4.2.2 DISTRIBUTION

Before human habitation, 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 populations once occurred around north-west Nelson and that prehistoric mainland colonies were genetically distinct from contemporary New Zealand sea lions, though became extinct shortly after the arrival of Polynesian settlers (Collins et al. 2014a, 2014b). Subsistence hunting on the mainland 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). Currently, most New Zealand sea lions are found in the New Zealand subantarctic, with individuals ranging to the New Zealand mainland and Macquarie Island. New Zealand sea lion breeding colonies[1] are highly localised, with most pups being born at two main breeding areas, the Auckland Islands and Campbell Island (Wilkinson et al. 2003, Chilvers 2008). At the Auckland Islands, there are three 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 Paradise Point, plus a small number of non-colonial breeders (Wilkinson et al. 2003, Chilvers 2008, Maloney et al. 2009, Maloney et al. 2012). Breeding on the Auckland Islands represents 71–87% of the pup production for the species, with the remaining 13–29% occurring on Campbell Island (based on concurrent pup counts in 2003, 2008 and 2010; see Section 0).

Although breeding is concentrated on the Auckland Islands and Campbell Island, births have also been reported from the Snares and Stewart Islands (Wilkinson et al. 2003, Chilvers et al. 2007), though there have been no recorded births of sea lions at the Snares Islands in 15 years (L. Chilvers, pers. comm.). Twenty-five sea lion pups were captured and tagged around Stewart Island during a DOC recreational hut and track maintenance trip in March 2012. Breeding success at the Stewart Island location has increased steadily since that time, with 41 pups tagged in 2017 (DOC & MPI 2017). Breeding is also taking place on the New Zealand mainland at the Otago Peninsula, originally the result of a single female arriving in 1992 and giving birth in 1993 (McConkey et al. 2002). Pup production at this location increased steadily, to 7 pups in 2013, followed by a more rapid increase, with 16 pups in 2017 (DOC & MPI 2017).

Annual pup counts at the four main breeding locations are reproduced in Figure 4.3.

On land, New Zealand sea lions are able to travel long distances and climb high hills, and are found in a variety of habitats including sandy beaches, grass fields, bedrock, and dense bush and forest (Gales 1995, Augé et al. 2012). Following the end of the females' oestrus cycle in late January, adult and sub-adult males disperse throughout the species' range, whereas dispersal of females (both breeding and non-breeding) are more restricted (Marlow 1975, Robertson et al. 2006, Chilvers & Wilkinson 2008).

4.2.3 FORAGING ECOLOGY

Foraging studies have been conducted on lactating female New Zealand sea lions from Enderby Island (Chilvers et al. 2005b, 2006, 2013, Chilvers & Wilkinson 2009), as well as at the Auckland Islands, Stewart Island, and the Otago Peninsula (see Augé et al. 2011a, 2014, Chilvers et al. 2011). Leung et al. (2012, 2013b, 2014b) have investigated foraging by juvenile New Zealand sea lions at Enderby

[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.'

Island, Auckland Islands 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 sea lion foraging at the Auckland Islands is currently in progress under MPI contract PRO2017-10, using all available satellite telemetry data to characterise diving behaviour and estimate overlap with fisheries. This research is scheduled to be completed in early 2019. Similar analyses of satellite-tracked individuals from Campbell Island is in preparation (M.-A. Lea, pers. comm.).Work

Previous analyses of sea lion foraging indicate that females from Enderby Island forage primarily within 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 showed 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 is spent submerged, with a mean dive depth of 130 ± 5 m (max. 597 m) and a mean dive duration of 4 ± 1 minutes (max. 14.5 minutes; Chilvers et al. 2006). Both juvenile (2–5 years old) female and male sea lions foraged to the north of the Auckland Islands, but mean distance travelled per foraging trip was shorter in females (99 ± 12 km, n = 19) compared to 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: overall mean dive depth 35 m), 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 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.

Studies conducted on female New Zealand sea lions suggest behavioural specialisation into one of two distinct foraging modes, i.e., benthic or meso-pelagic (Chilvers & Wilkinson 2009). Alternatively, these behaviours may represent two extremes of a continuum rather than distinct modes (Roberts et al. in prep). Benthic divers have fairly consistent dive profiles, reaching similar depths (120 m on average) on consecutive dives in relatively shallow water to presumably feed on benthic prey. Meso-pelagic divers, by contrast, exhibit more varied dive profiles, undertaking both deep (over 200 m) and shallow (less than 50 m) dives over deeper water. Benthic divers tend to forage further from their breeding colonies, making their way to the north-eastern limits of Auckland Islands' shelf, whereas meso-pelagic divers tend to forage along the north-western edge of the shelf over depths of approximately 3000 m (Chilvers & Wilkinson 2009). Meynier et al. (2014), employing fatty acid (FA) analyses of blubber samples, found that FA profiles were different in benthic-diving and meso-pelagic-diving lactating New Zealand sea lions suggesting a different utilisation of prey resources, and that while prey species taken were similar across both dive types, the proportion of particular prey differed between the two dive categories. Further, Meynier et al. (2014) found that the body condition index (BCI: 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 compared to benthic divers.

The differences in dive profiles have further implications for the animals' 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 in order to aerobically metabolise the lactate before they can undertake subsequent dives. Chilvers et al. (2006) estimated that lactating female New Zealand sea lions 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). New Zealand 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). Any observed differences may reflect differences in habitat (including prey availability) comparing 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 to Otago. New evidence from satellite tracked individuals at the Campbell Islands (M-A Lea, pers. comm.) and from analysis of sea lion prey including a dedicated ocean survey (Roberts et al. in prep) suggest that sea lions at the Subantarctic Islands may sometimes suffer from low prey availability, and may be forced to forage at the limits of their physiological capabilities. This would make these populations particularly susceptible to environmental variability affecting availability of preferred prey. Roberts and Doonan (2016) identify nutritional stress as a potentially significant threat to sea lions.

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

Characteristic	Auckland Islands	Otago
Reproduction at age 4	< 5% of females	> 85% of females
Average mass at 8–13 years of age	112 kg	152 kg
Foraging distance from shore	102.0 ± 7.7 km (max = 175 km)	4.7 ± 1.6 km (max = 25 km)
Time spent foraging at sea	66.2 ± 4.2 hrs	11.8 ± 1.5 hrs
Dive depth	129.4 ± 5.3 m (max = 597 m)	20.2 ± 24.5 m (max = 389 m)
Dives estimated to exceed ADL	68.7 ± 4.4%	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 royal albatross *Diomedea epomophora*), elasmobranchs (rough skate *Raja nasuta*), teleost fish (e.g., opalfish *Hemerocoetes* spp., hoki *Macruronus novaezelandiae*, red cod *Pseudophycis bachus*, jack mackerel *Trachurus* spp., barracouta *Thyrsites atun*), 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). 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 bycaught animals tend to be biased towards the target species of the fishery concerned (e.g., squid in the SQU 6T 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). This approach 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, while 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 baracoutta 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., opalfish, 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 were used to estimate trophic histories over an extended historical period. Graham (in prep) analysed 292 samples from the annual bands found in 17 sea lion teeth. This dataset represents the nitrogen and carbon isotopic histories of 13 female sea lions dating from 1935 to 2005, along with the histories of 4 males that were analysed initially to refine methods and techniques. Nitrogen isotope ($\delta^{15}N$) data indicate an animal's trophic ecology and changes in their foraging strategies. First, the male sea lions consistently forage at a higher trophic level than the females. The $\delta^{15}N$ values of the remaining 13 females shows 3 distinct features. At a broad scale, there is considerable variation between individuals, suggesting variable foraging strategies. Second, a maternal or lactation signal was observed in almost all teeth samples, indicating that teeth were sampled correctly. 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 not present in any but 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 $\delta^{13}C$ 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 $\delta^{13}C$ 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 $\delta^{13}C$ 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) or b) the females shifted their main foraging strategy (e.g., benthic vs. mesopelagic related to available prey). More samples will be required, especially in the earlier time period to resolve the timing of these isotopic changes as it relates to estimated changes in fish stocks. In addition, examining other species in the region during this time period will enable larger-scale assessments and how they might relate to shifts in ocean conditions.

4.2.4 REPRODUCTIVE BIOLOGY

New Zealand sea lions exhibit marked sexual dimorphism, with adult males being larger and darker in colour 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, but Childerhouse et al. (2010b) reported a maximum estimated age for females of 28 years (although the AEWG had some concerns about the methods used and this estimate may not be reliable). Females can become sexually mature as early as age two and give birth the following year. However, at the Auckland Islands most do not breed until they are six years old (Childerhouse et al. 2010b); at Otago Peninsula most females breed by age four (Roberts & Doonan 2016). Males generally reach sexual maturity at four years of age, 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 (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 three 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 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).

Pups' 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 10 days after giving birth before alternating between foraging trips lasting approximately two days out at sea and returning for about one day to suckle their pups (Gales & Mattlin 1997, Chilvers et al. 2005b). New Zealand 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 BCI 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).

4.2.5 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). Since 1995, the Department of Conservation (DOC) has conducted mark-recapture counts at each of the main breeding colonies at the Auckland Islands 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; Figure 4.3 and Table 4.2), with the largest single-year decline (31%) occurring between the 2008 and 2009 counts.

Since 2009 the breeding population at the Auckland Islands appears to have stabilised and may be recovering, but it is too early to discern a definite trend. The most recent estimate of pup production for the Auckland Islands population was 1965 pups in 2017 of which 349 were counted at Sandy Bay and 1549 were counted at Dundas Island, using the mark-recapture method (Table 4.2 and Figure 4.3).

The steep decline in total pup production on the Auckland Islands seen from 1997–98 until 2008–09 appears to have ended. Since the lowest-ever record of total pup production at the Auckland Islands in 2008–09, pup production has seen annual increases in six of the last eight years, and overall production in 2017 was the highest seen since 2009. While the apparent stabilisation of total pup production is a positive step, it is important to note that pup production is still substantially lower than the peak in 1997–98 (Childerhouse et al. 2016). The future population trajectory remains highly uncertain, depending on what period of time is used to estimate future demographic rates in the forward projections.

The population models used to inform the TMP (Roberts & Doonan 2016) used demographic rates from the past 20 years, which includes the period of steepest population decline; these models suggest that the total Auckland Island population will continue to decline (see Section 4.4.3.4, Roberts & Doonan 2016). However projections based on demographic rates from a 30-year or 10-year window would be expected to produce different trajectories. Because it is not possible to anticipate what environmental or demographic conditions are likely to prevail in the future, uncertainty of this nature is best addressed with model sensitivities.

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, incidental fisheries captures), as well as data on the resighting of tagged animals and pup production estimates (Table 4.3).

For the Campbell Island population, the latest estimate of pup production was 696 pups in 2015 (Childerhouse et al. 2015a). Estimates of pup production at Campbell Island increased sharply 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. 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). Early pup mortality at Campbell Island has been relatively high in all recent census years, including: 1998 (31%), 2003 (36%), 2008 (40%), 2010 (55%) and 2015 (58%, the highest recorded at any New Zealand sea lion breeding site) (see Childerhouse et al. 2005, 2015a, Maloney et al. 2009, 2012, McNally et al. 2001).

For the Otago Peninsula site, annual pup production has ranged from 0 to 16 pups since the 1994–95 breeding season, with 16 recorded in 2016–17 (Figure 4.3). Sea lions at Otago are of special interest because they highlight the potential for establishing new breeding colonies, in this case originating with a single pregnant female (McConkey et al. 2002). The TMP identifies that the viability of new colony locations on the New Zealand mainland is of particular importance for the restoration of New Zealand sea lions to non-threatened status.

Sea lions have also established at Stewart Island, and pup census estimates have also 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 41 pups in 2017 (Chilvers 2014, Roberts & Doonan 2016; Figure 4.3).

Established anthropogenic sources of mortality in New Zealand sea lion include: historic subsistence hunting and commercial harvest (Gales 1995, Childerhouse & Gales 1998); pup entrapment in rabbit burrows prior to rabbit eradication from Enderby Island in 1993 (Gales & Fletcher 1999); human disturbance, including attacks by dogs, vehicle strikes and deliberate shooting on mainland New Zealand (Gales 1995); and incidental captures in fisheries (see Section 4.4).

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).

Pupping season	Annual pup census estimate						
	Auckland Islands			Campbell Island	Otago Peninsula	Catlins	Stewart Island
	Dundas	Sandy Bay	All				
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		
2005	1 587	441	2 148		4		
2006	1 581	422	2 089		6	1	
2007	1 693	437	2 224		3	1	
2008	1 635	448	2 175	583	5	1	
2009	1 132	301	1 501		4	1	
2010	1 369	385	1 814	681	5	1	
2011	1 089	378	1 550		5	1	16
2012	1 248	361	1 684		4	2	25
2013	1 491	374	1 940		5	1	26
2014	1 213	290	1 575		3	1	32
2015	1 230	286	1 576	696	8	0	36
2016	1 347	321	1 727		11	4	31
2017	1 549	349	1 965		12	4	41

Table 4.3: Pup production and population estimates of New Zealand sea lions from the Auckland Islands from 1995 to 2017. Pup production data are direct counts or mark-recapture estimates from Chilvers et al. (2007), Robertson and 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. Population estimates from P.A. Breen, estimated in the model by Breen et al. 2016. Year refers to the second year of a breeding season (e.g., 2010 refers to the 2009–10 season).

Year	Pup production estimate		Population size estimate	
	Mean	Standard error (for mark recapture estimates)*	Median	90% confidence interval
1995	2 518	21	15 675	14 732–16 757
1996	2 685	22	16 226	15 238–17 318
1997	2 975	26	16 693	15 656–17 829
1998	3 021	94	16 911	15 786–18 128
1999	2 867	33	15 091	13 932–16 456
2000	2 856	43	15 248	14 078–16 586
2001	2 859	24	15 005	13 870–16 282
2002	2 282	34	13 890	12 856–15 079
2003	2 518	38	14 141	13 107–15 295
2004	2 515	40	14 096	13 057–15 278
2005	2 148	34	13 369	12 383–14 518
2006	2 089	30	13 110	12 150–14 156
2007	2 224	38	13 199	12 231–14 215
2008	2 175	44	12 733	11 786–13 757
2009	1 501	16	12 065	11 160–13 061
2010	1 814	36		
2011	1 550[2]	41		
2012	1 684	22		
2013**	1 940	50		
2014**	1 575	19		
2015	1 576		***	
2016	1 727			
2017	1 965			

* Calculated as the sum of standard errors associated with estimates for Sandy Bay and Dundas (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.

In addition to the established effects, there are a number of other anthropogenic effects that may influence New Zealand sea lion mortality. However, their role, if any, is presently unclear. These include: possible competition for resources between New Zealand sea lions and the various fisheries (Robertson & Chilvers 2011, Bowen 2012, Roberts et al. in prep); 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 eco-tourism.

Other sources of mortality include epizootics, particularly *Campylobacter* that 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), and *Klebsiella pneumoniae* that killed 33% and 21% of diagnosed 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 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

[2] Due to extreme weather conditions there was some delay in making the 2010–11 pup count, which may affect comparability with previous years. However DOC's analysis suggests any such effect is unlikely to be large (Chilvers & Wilkinson 2011).

1998–99 to 2004–05 and were still occurring up to the end of sampling (Castinel et al. 2007, Roe 2011). The 1998 epizootic event may have affected the fecundity of the surviving pups, reducing their breeding rate relative to other cohorts (Gilbert & Chilvers 2008), though their 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). There are also occurrences of 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), drowning in wallows and male aggression towards females and pups (Wilkinson et al. 2000, Chilvers et al. 2005a).

Despite a historic reduction in population size 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).

4.2.6 RELATING DEMOGRAPHIC RATES TO DRIVERS OF POPULATION CHANGE

Demographic assessments have been conducted to identify the proximate demographic causes of population decline at the Auckland Islands. An assessment using mark-resighting data from the Enderby Island sub-population yielded estimates of average annual survival for prime-age females of 0.90 for females that did not breed and 0.95 for females that did breed (MacKenzie 2011). In another assessment, state space demographic models fitted to pup production estimates, age distribution observations and a long time series of mark-resighting observations were developed using NIWA's demographic modelling software SeaBird to estimate year-varying survival, probability of pupping and age-at-first-pupping (Roberts et al. 2014a). This study concluded that low pupping rates (including occasional years with very low rates), a declining trend in cohort survival to age two and to age five since the early 1990s, and relatively low adult survival (age 6–14) from 1999–2000 to 2010–11 are responsible for declining pup production at Sandy Bay from the late 1990s to 2009. In addition, very low pup survival estimates were obtained for all years since 2004–05, which will compromise breeder numbers and pup production resulting from births at Sandy Bay in the immediate future (Figure 4.4) (Roberts et al. 2014a).

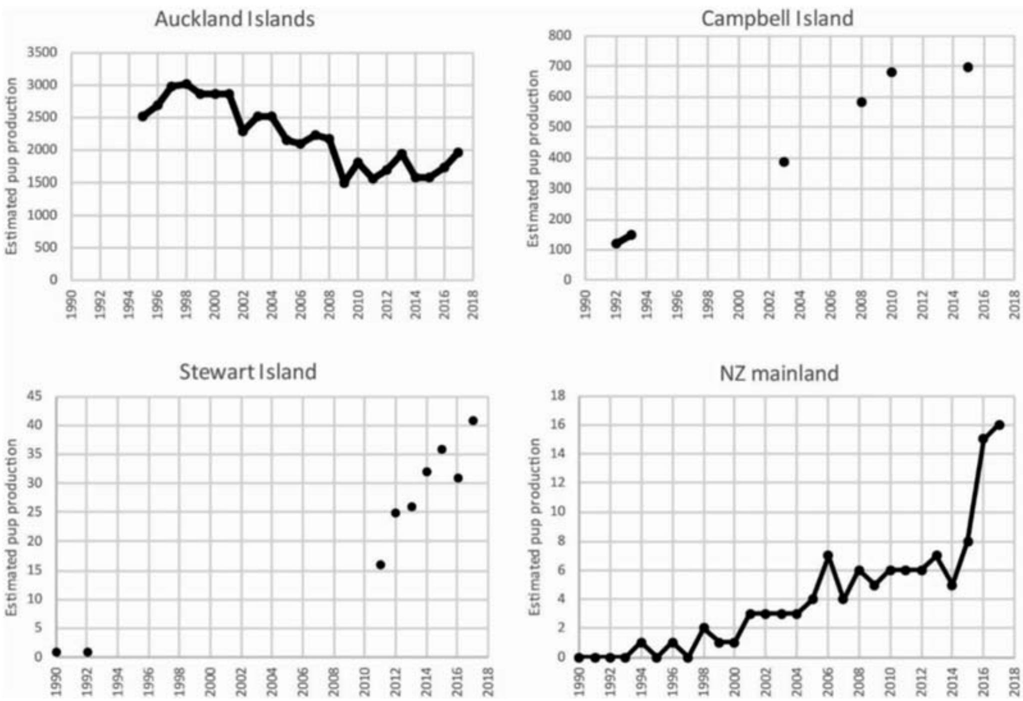

Figure 4.3: Annual sea lion pup count estimates from breeding sites. Note that the scale for each figure is different (DOC & MPI 2017, adapted from Roberts & Doonan 2016).

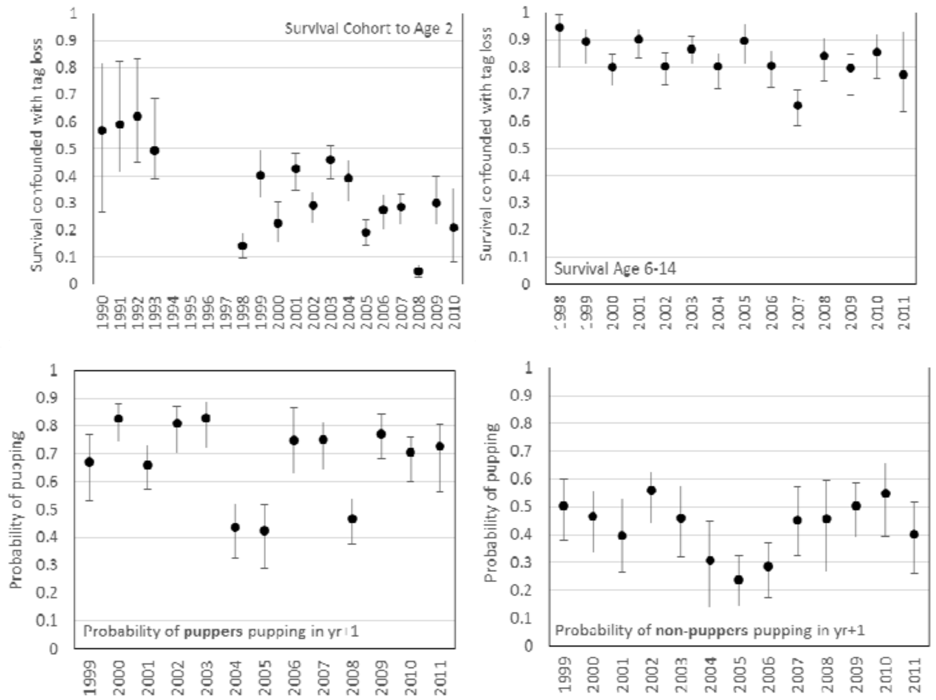

Figure 4.4: Annual estimates of cohort-specific survival to age 2 (top left), survival at age 6–14 (top right), probability of puppers pupping (bottom right) and non-puppers pupping (bottom right) of female New Zealand sea lions at Sandy Bay; survival estimates confounded with tag loss and likely to be lower than true values; points are median estimates, bars are 95% c.i. (Roberts et al. 2014a).

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 2016).

Climate indices including Inter-decadal 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 prior 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 (juveniles) or ages 6–14 (adults) were not well correlated with estimated captures or interactions in the Auckland Islands southern arrow squid trawl fishery (SQU 6T) and estimated captures are relatively low in other commercial fisheries around the Auckland Islands (Thompson et al. 2011). However, from 1998–99 to 2003–04 survival at ages 6–14 was negatively correlated with the survival of pups born in the previous year, consistent with the high energetic costs of lactation compromising maternal survival.

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 is likely that these effects are not independent, as nutritional stress can be expected to predispose the population to higher rates of disease mortality.

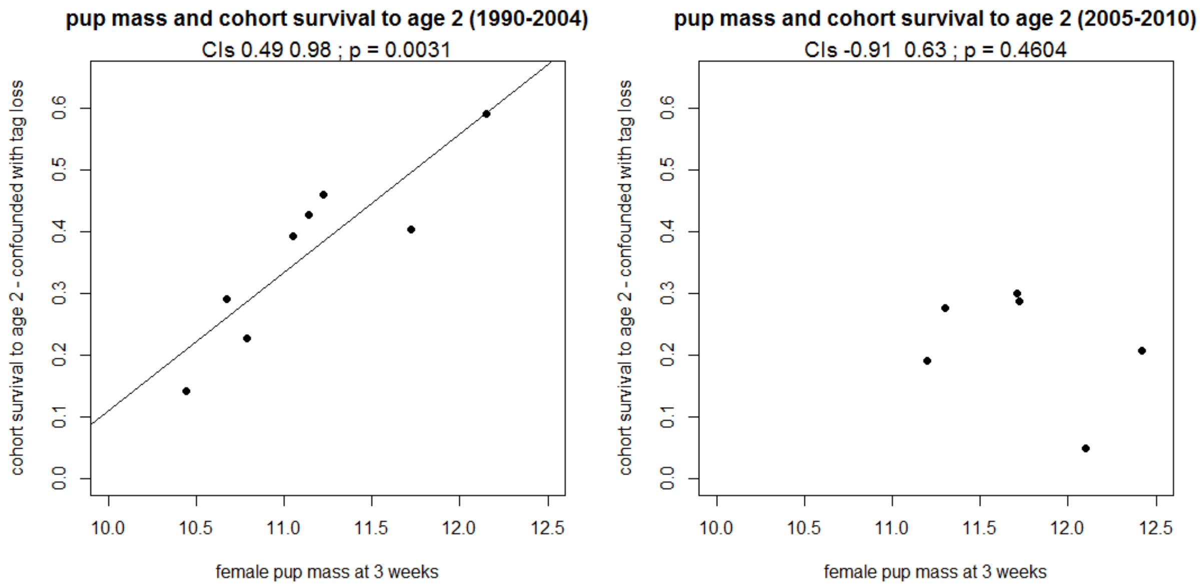

Figure 4.5: Pup mass of females and demographic modelling estimate of cohort survival to age 2; survival estimates confounded with tag loss rate; regression line shown for correlations significant at the 5% level.

4.2.7 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2010). The risk of extinction for 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 2008, the IUCN updated the Red List status of New Zealand sea lions, listing them as Vulnerable, A3b[3] on the basis of a marked (30%) decline in pup production at the Auckland Islands in the previous 10 years, at some of the major rookeries (Gales 2008). The IUCN further recommended that the species should be reviewed within a decade in light of what they considered to be the current status of New Zealand sea lions (i.e., declining pup production, reducing population size, severe disease outbreaks).

In 2010, DOC updated the New Zealand Threat Classification status of all New Zealand marine mammals (Baker et al. 2010). In the revised list, New Zealand sea lions had their threat classification increased from At Risk, Range Restricted[4] to Nationally Critical under criterion C[5] with a Range Restricted qualifier based on the recent rate of decline at the Auckland Islands (Baker et al. 2016). The New Zealand Threat Classification status for New Zealand sea lions will be updated in early 2018.

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[3] 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% 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)).

[4] 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 sub-populations (Townsend et al. 2008).

[5] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of sub-populations) 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).

4.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Reviews of fisheries interactions among pinnipeds globally can be found in Read et al. (2006), Woodley & Lavigne (1991), 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 subantarctic waters (Figure 4.6); particularly the Auckland Islands arrow squid fishery (SQU 6T), but also the Auckland Islands scampi fishery (SCI 6A), other Auckland Islands trawl fisheries, the Campbell Island southern blue whiting (*Micromesistius australis*) fishery (SBW 6I) and the Stewart-Snares Shelf fisheries targeting mainly arrow squid (SQU 1T; Thompson & Abraham 2010, Thompson et al. 2011, 2013).[6] New Zealand sea lions forage to depths of up to 600 m and overlap with trawling at up to 500 m depth for arrow squid, 250–600 m depth for spawning southern blue whiting, and 350–550 m depth for scampi (Tuck 2009, Ministry of Fisheries 2011). There is seasonal variation in the distribution overlap between New Zealand sea lions and the target species fisheries (Table 4.4) Breeding male sea lions in the Auckland Island 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 Island area year-round, ashore to give birth for up to 10 days during December and January and then dividing their time between foraging at sea (~2 days) and suckling their pup ashore (~1.5 days; Chilvers et al. 2005b). The SQU 6T fishery currently operates between February and July, 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 may operate at any time of the year but does not operate continuously.

4.4.1 QUANTIFYING FISHERIES INTERACTIONS

Incidental captures of New Zealand sea lions are recorded by government observers. For those fisheries in which the majority of historical captures have been observed, observer coverage levels have increased substantially in recent years (Tables 4.5-4.8). For example in the SQU 6T fishery, observer coverage ranged 28–45% from 2002–03 to 2011–12, but increased to 85–92% in the four years since 2012-13 (Table 4.5). Similarly, the Campbell Island southern blue whiting fishery has had 100% observer coverage since 2012-13 (Table 4.8Table 4.5).

Observed New Zealand sea lion captures are used in models to estimate total captures across the entire fishing fleet in each fishing year (Smith and Baird 2007b, Thompson and Abraham 2010, Abraham and Thompson 2011, Abraham & Berkenbusch 2017). This approach is currently being applied using information collected under DOC project INT2014-01 and analysed under MPI project PRO2013-01. Estimates for the SQU 6T and Campbell Island fisheries are generated using Bayesian models, whereas those for Auckland Islands scampi fisheries, other Auckland Islands trawl fisheries, and the Stewart-Snares Shelf fisheries are produced using ratio estimates (see Tables 4.6–4.8, and detailed information in Thompson et al. 2013, Abraham & Berkenbusch 2017).

[6] See the Report from the Fisheries Assessment Plenary, May 2011 (Ministry of Fisheries 2011) for further information regarding the biology and stock assessments for these species.

Table 4.4: Monthly distribution of New Zealand sea lion activity and the main trawl fisheries with observed reports of New Zealand sea lion incidental captures (see text for details).

<table>
<tr><th>New Zealand sea lions</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th></tr>
<tr><td>Breeding males</td><td colspan="2">Dispersed at sea or at haulouts</td><td colspan="3">At breeding colony</td><td colspan="7">Dispersed at sea or at haulouts</td></tr>
<tr><td>Breeding females</td><td colspan="3">At sea</td><td colspan="2">At breeding colony</td><td colspan="7">At breeding colony and at-sea foraging and suckling</td></tr>
<tr><td>New pups</td><td colspan="3"></td><td colspan="9">At breeding colony</td></tr>
<tr><td>Non-breeders</td><td colspan="12">Dispersed at sea, at haulouts, or breeding colony periphery</td></tr>
<tr><th>Major fisheries</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th></tr>
<tr><td>Hoki trawl</td><td></td><td colspan="8">Chatham Rise and Stewart-Snares Shelf</td><td colspan="3">Cook Strait, west coast South Island, Puysegur</td></tr>
<tr><td>Squid</td><td colspan="3"></td><td colspan="2">Stewart-Snares Shelf</td><td colspan="6">Auckland Islands and Stewart-Snares Shelf</td><td></td></tr>
<tr><td>Southern blue whiting</td><td colspan="2">Pukaki Rise and Campbell Rise</td><td colspan="9"></td><td>Bounty Islands</td></tr>
<tr><td>Scampi</td><td colspan="12">Auckland Islands</td></tr>
</table>

Captures include sea lions captured in nets and brought on deck (both dead and alive). Captures necessarily exclude the unknown fraction of animals that exit trawls through Sea Lion Exclusion Devices (SLEDs), as well as bodies that are recovered in a decomposed state at the time of capture, and individuals that climb aboard the vessel of their own volition (Smith & Baird 2007b, Thompson & Abraham 2010, Thompson et al. 2013).

Interactions are defined as the number of sea lions that enter the net and would have been captured if no SLED had been used (i.e., in the SQU 6T fishery). Interactions are estimated using a statistical model fitting to observed capture rates both before and after the deployment of SLEDs, with an additional term to estimate SLED efficacy, i.e. the proportion of interactions in which the sea lion escapes via the SLED (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.

Models suggest that the interaction rate (i.e. number of interactions per 100 trawl tows) for female New Zealand sea lions at the Auckland Islands is influenced by a number of factors, including year, distance from the rookery, tow duration, and change of tow direction (Smith & Baird 2005). Conversely, the interaction rate of male New Zealand sea lions is influenced by year, the number of days into the fishery (males leave the rookeries soon after mating whereas females remain with the pups), and time of day (Smith & Baird 2005).

In the years since SLEDs were introduced to some vessels in the SQU 6T fishery in 2001–02, both the observed and estimated numbers of New Zealand sea lion captures have declined (Table 4.6). The same trend is present in the mean estimated number of interactions, however these estimates have become increasingly uncertain because since the universal adoption of SLEDs in 2006–07, the interaction rate is confounded in the model with an unknown SLED efficacy. In the most recent models, interaction rate estimates are effectively unbounded, and model estimates in particular years are unstable as new years' data are added. For this reason, from 2017 MPI will cease relying on new estimates of interaction rates generated using data from the period after SLEDs were in universal use.

Observed and estimated New Zealand sea lion captures and capture rates in the Campbell Island southern blue whiting fishery have been highly variable (Table 4.8). Following the 2012–13 season in which 21 captures were observed, SLEDs were voluntarily introduced to that fishery. Since that time, 100% of tows in these fisheries have been observed, and annual captures have ranged from 2 to 6 sea lions. For the Auckland Islands scampi and other target fisheries, and the Stewart-Snares Shelf trawl fisheries, the observed and estimated numbers of New Zealand sea lion captures have fluctuated without trend (Table 4.7).

4.4.2 MANAGING FISHERIES INTERACTIONS

For New Zealand sea lions, efforts to mitigate incidental captures in fisheries have focused on the SQU 6T fishery. From 2017, advice to manage sea lion interactions in this fishery has been developed in consultation with the Squid 6T Operational Plan Technical Advisory Group, including representatives from government and stakeholder groups as well as technical experts and advisors. Under the present Operational Plan, adopted in December 2017, MPI sets a fishing-related mortality limit (FRML) for sea lions in the Auckland Islands squid trawl fishery (SQU 6T) 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 current PST is 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.

Present management also includes previously adopted measures including spatial closures, voluntary measures promoting the use of Sea Lion Exclusion Devices (SLEDs) and other voluntary measures. In 1982 the Minister of Fisheries established a 12 nautical mile exclusion zone around the Auckland Islands from which all fishing activities were 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.

SLEDs were first utilised on some vessels in the SQU 6T fishing fleet in 2001–02 (Figure 4.7). SLED use increased in subsequent years. The use of SLEDs is not mandatory, but use of a certified SLED is required by the current industry body (the Deepwater Group) and is necessary to receive the 'Discount Rate' relative to the tow limit applied by MPI (see section 4.4.2.2, below). For these reasons, from 2006–07 a standardised model Mark 13/3 SLED has been universally employed by all vessels in the SQU 6T fleet. SLED deployment is monitored and audited by MPI observers.

In 1992, the Ministry adopted a fisheries-related mortality limit (FRML; previously referred to as a maximum allowable level of fisheries-related mortality or MALFiRM) 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 will be closed for the remainder of the season.

The original 'MALFiRM' was calculated using the potential biological removal approach (PBR; Wade 1998) and was used from 1992–93 to 2003–04 (Smith & Baird 2007a). Since 2003–04 the FRML has been translated into a maximum permitted number of tows calculated from assumed interaction and SLED efficacy rates, 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 are no longer a reliable index of the number of sea lions interacting with the net, and there is uncertainty about the survival rate of sea lions exiting the net via the SLED ('SLED efficacy'); for this reason the number of sea lion deaths from fishery interactions cannot be observed directly.

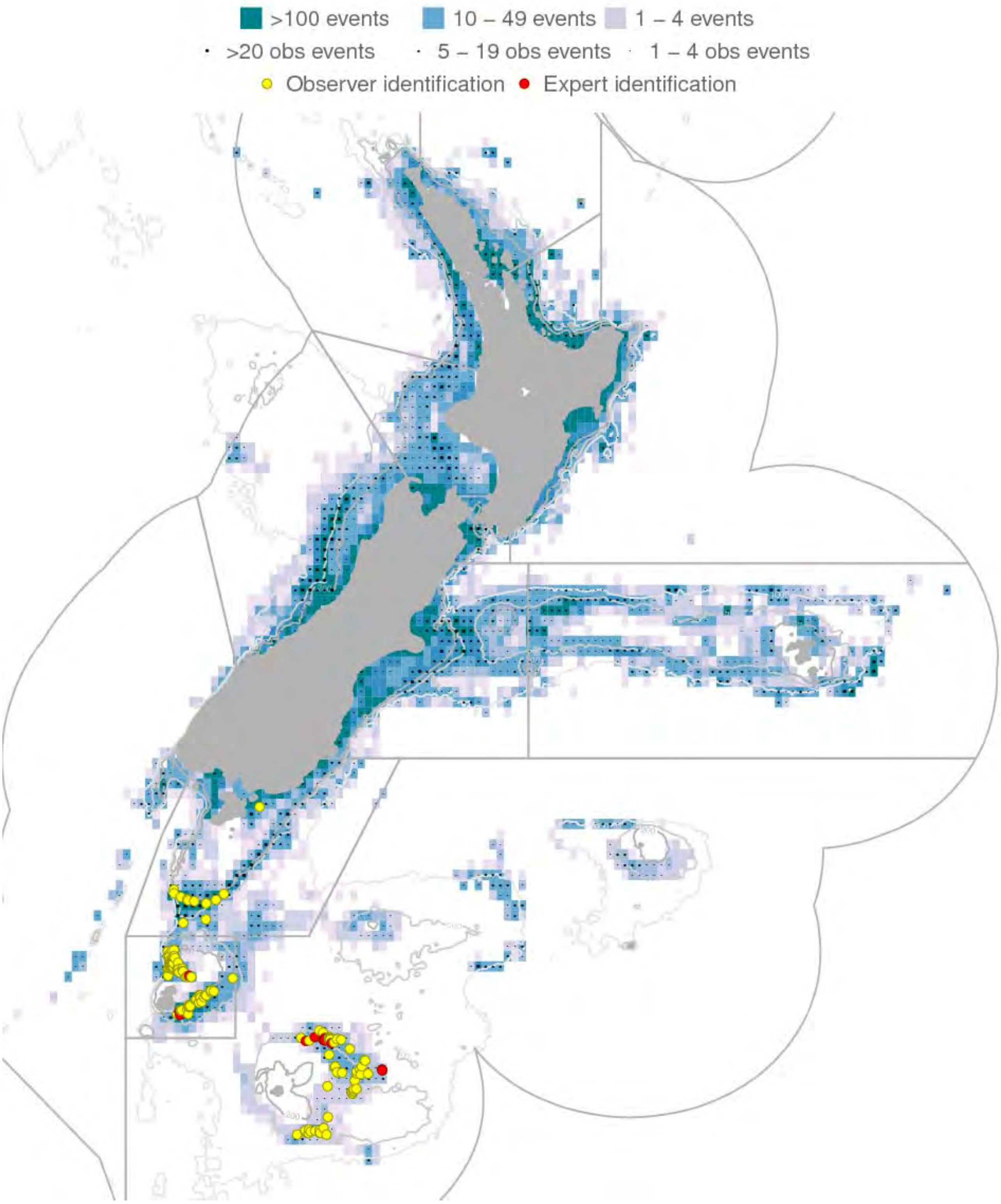


Figure 4.6: Distribution of trawl fishing effort and observed New Zealand sea lion captures, 2002–03 to 2015–16 (http://data.dragonfly.co.nz/psc. Data version v2016001). Fishing effort is mapped into 0.2-degree cells, with the colour of each cell indicating the amount of effort (number of fishing events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing is only shown if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell.

Table 4.5: Sea lion captures in all commercial trawl fisheries in New Zealand's Exclusive Economic Zone between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). 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, ratio estimate, or both combined), the number of estimated sea lion captures, and estimated interactions (with 95% confidence intervals, c.i.). Interactions are defined as the number of sea lions that would have been caught if no Sea Lion Exclusion Devices (SLEDs) had been used (see Thompson et al. 2013 and 2016 for details).

Fishing year	Fishing effort		Observed captures		Estimated captures			Estimated interactions	
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.	Mean	95% c.i.
2002–03	130 175	5.25	12	0.175	Both	31	21–44	59	36–93
2003–04	120 847	5.42	21	0.321	Both	58	41–79	225	122–402
2004–05	120 464	6.40	14	0.181	Both	50	33–72	187	95–344
2005–06	109 951	6.02	15	0.227	Both	49	33–70	175	87–328
2006–07	103 327	7.68	12	0.151	Both	42	27–60	119	57–245
2007–08	89 536	10.11	11	0.122	Both	30	19–43	178	40–823
2008–09	87 550	11.43	3	0.030	Both	19	10–32	146	25–686
2009–10	92 889	10.26	15	0.157	Both	44	30–63	197	52–847
2010–11	86 091	8.81	6	0.079	Both	27	16–40	113	27–519
2011–12	84 424	11.06	1	0.011	Both	12	5–20	70	12–326
2012–13	83 832	14.79	25	0.202	Both	32	27–39	120	51–437
2013–14	85 113	15.51	4	0.030	Both	10	6–17	69	18–255
2014–15	78 754	17.61	8	0.058	Both	12	8–17	81	27–281
2015–16	78 040	18.17	4	0.028	Both				

Table 4.6: Sea lion captures in the Auckland Islands squid trawl fishery between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). 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, ratio estimate, or both combined), the number of estimated sea lion captures, estimated interactions, and estimated strike rate (with 95% confidence intervals, c.i.). Interactions are defined as the number of sea lion that would have been caught if no Sea Lion Exclusion Devices (SLEDs) had been used (see Thompson et al. 2013 and 2016 for details).

Fishing year	Fishing effort		Observed captures		Estimated captures			Estimated interactions		Estimated strike rate	
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.	Mean	95% c.i.	Mean	95% c.i.
2002–03	1 466	28.38	11	2.644	Model	18	12–28	47	25–79	3.1	1.9–4.9
2003–04	2 594	30.57	16	2.018	Model	39	26–59	206	104–383	7.8	4.0–14.0
2004–05	2 693	29.93	9	1.117	Model	30	16–49	167	76–323	6.1	2.8–11.7
2005–06	2 459	22.37	10	1.818	Model	26	15–43	153	65–306	6.1	2.7–12.3
2006–07*	1 317	40.70	7	1.306	Model	15	9–25	93	33–216	6.8	2.4–15.6
2007–08	1 265	46.72	5	0.846	Model	12	6–22	160	24–804	9.4	1.8–39.8
2008–09	1 925	39.64	2	0.262	Model	7	2–15	134	14–672	5.3	0.7–24.5
2009–10	1 188	25.51	3	0.990	Model	12	5–26	165	22–818	10.8	1.9–44.8
2010–11	1 583	34.55	0	0.000	Model	3	0–10	90	5–501	4	0.3–17.9
2011–12	1 281	44.57	0	0.000	Model	2	0–6	60	3–319	3.5	0.3–16.5
2012–13	1 027	86.17	3	0.339	Model	4	3–6	73	8–384	5.3	0.8–24.5
2013–14	737	84.40	2	0.322	Model	2	2–4	47	5–231	5.0	0.7–22.8
2014–15	633	88.31	1	0.179	Model	1	1–3	44	3–236		
2015–16	1 367	92.25	0	0	Model						

* SLEDs standardised and in widespread use.

Table 4.7: Sea lion captures in trawl fisheries targeting scampi and targeting other species adjacent to the Auckland Islands between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). 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 number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details).

Fishing year	Fishing effort		Observed captures		Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Auckland Islands scampi							
2002–03	1 351	11.10	0	0.000	Ratio	7	2–15
2003–04	1 363	12.40	3	1.775	Ratio	10	5–18
2004–05	1 275	0.00	0	0.000	Ratio	8	2–16
2005–06	1 331	8.87	1	0.847	Ratio	8	3–16
2006–07	1 328	7.61	1	0.990	Ratio	8	3–16
2007–08	1 327	7.01	0	0.000	Ratio	8	2–15
2008–09	1 457	4.19	1	1.639	Ratio	10	3–18
2009–10	NA	NA	0	0.000	Ratio	5	1–11
2010–11	1 401	14.78	0	0.000	Ratio	7	2–15
2011–12	NA	NA	0	0.000	Ratio	7	2–14
2012–13	1 093	12.44	0	0.000	Ratio	6	1–12
2013–14	NA	NA	0	0.000	Ratio	5	1–11
2014–15	NA	NA	0	0.000	Ratio	3	0–8
2015–16	1 414	4.7	0	0.000			
Auckland Islands other							
2002–03	543	13	0	0.000	Ratio	2	0.0–1.1
2003–04	289	17	0	0.000	Ratio	1	0.0–1.0
2004–05	170	7	0	0.000	Ratio	1	0.0–1.8
2005–06	39	15	0	0.000	Ratio	0	0.0–2.6
2006–07	38	5	0	0.000	Ratio	0	0.0–2.6
2007–08	147	45	0	0.000	Ratio	0	0.0–1.4
2008–09	121	50	0	0.000	Ratio	0	0.0–0.8
2009–10	77	66	0	0.000	Ratio	0	0.0–1.3
2010–11	131	37	0	0.000	Ratio	0	0.0–1.5
2011–12	57	30	0	0.000	Ratio	0	0.0–1.8
2012–13	60	43	0	0.000	Ratio	0	0.0–1.7
2013–14	203	23	0	0.000	Ratio	1	0.0–1.0
2014–15	224	31	0	0.000	Ratio	1	0.0–1.3

Table 4.8: Sea lion captures in Campbell Island southern blue whiting (SBW) and in Stewart-Snares Shelf trawl fisheries between 2002 and 2016 (http://data.dragonfly.co.nz/psc. Data version v2017001). 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 number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details). [Continued on next page]

Fishing year	Fishing effort		Observed captures		Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Campbell Island SBW							
2002–03	599	43	0	0	Model	1	0–3
2003–04	690	34	1	0.4	Model	3	1–9
2004–05	726	37	2	0.7	Model	5	2–13
2005–06	521	28	3	2.1	Model	10	3–22
2006–07	544	32	6	3.5	Model	15	6–30
2007–08	557	41	2	0.9	Model	8	5–14
2008–09	627	20	0	0	Model	1	0–7
2009–10	550	43	11	4.7	Model	24	15–37
2010–11	886	39	6	1.7	Model	15	8–25
2011–12	592	77	0	0	Model	1	0–4
2012–13	693	100	21	3	Model	21	21–21
2013–14	588	100	2	0.3	Model	2	2–2

Table 4.8 [Continued]:

Fishing year	Fishing effort		Observed captures		Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Campbell Island SBW							
2014–15	566	100	6	1.1	Model	6	6–6
2015-16							
Stewart-Snares (mainly squid)							
2002–03	17 104	6.68	0	0.000	Ratio	3	0–7
2003–04	16 441	7.49	1	0.082	Ratio	4	1–8
2004–05	17 273	10.81	3	0.161	Ratio	7	3–11
2005–06	15 662	7.22	1	0.089	Ratio	4	1–8
2006–07	15 008	9.00	1	0.074	Ratio	3	1–6
2007–08	12 504	12.34	1	0.065	Ratio	3	1–6
2008–09	11 063	15.51	0	0.000	Ratio	1	0–4
2009–10	12 435	15.91	1	0.051	Ratio	2	1–5
2010–11	10 758	12.42	0	0.000	Ratio	1	0–4
2011–12	11 839	15.79	1	0.053	Ratio	2	1–4
2012–13	11 625	26.19	1	0.033	Ratio	2	1–4
2013–14	11 289	21.86	0	0.000	Ratio	1	0–3
2014–15	10 009	24.18	1	0.041	Ratio	1	1–3
2015–16	9 597	21.13	1	0.049	Ratio		

*SLEDs introduced in that year.

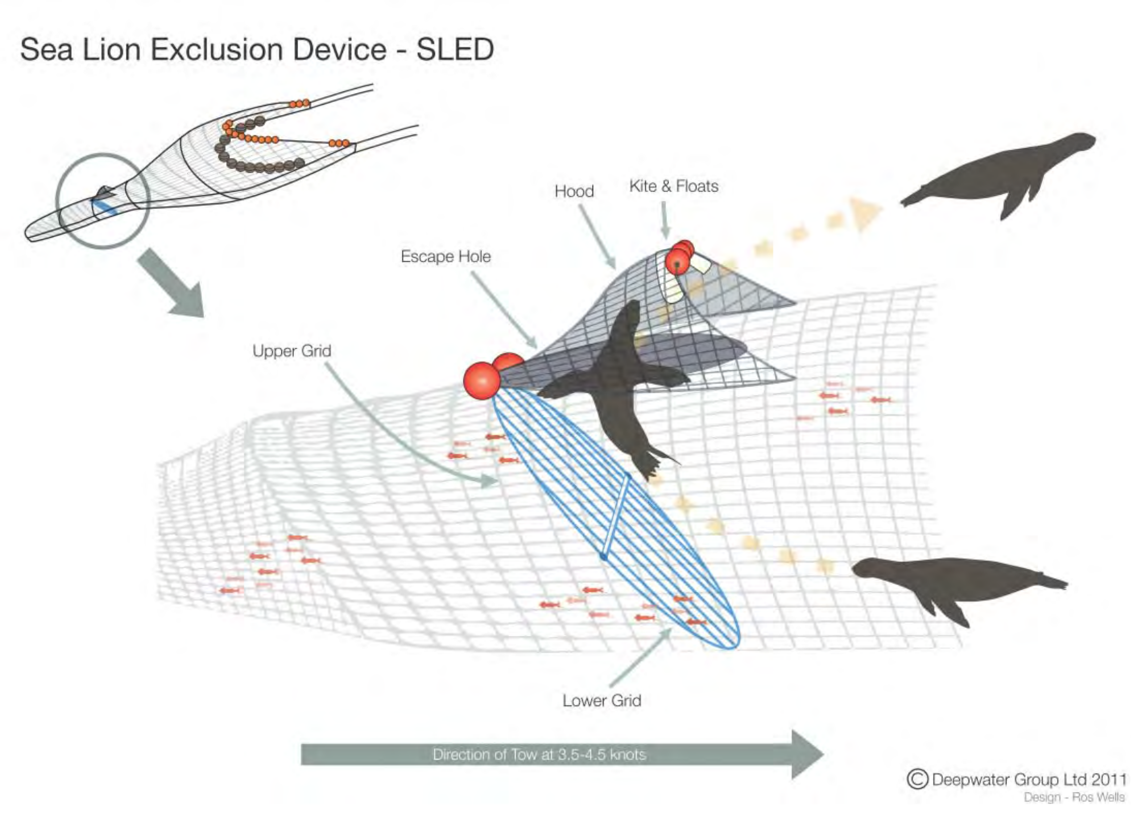


Figure 4.7: Diagram of a New Zealand sea lion exclusion device (SLED) inside a trawl net. Image courtesy of the Deepwater Group.

4.4.2.1 SEA LION EXCLUSION DEVICE (SLED) DEVELOPMENT AND USE

In 2004, the Minister of Fisheries requested that the squid fishery industry organisation (Squid Fishery Management Company), government agencies and other stakeholders with an interest in sea lion conservation 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; Clement and Associates Ltd. 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 mid-water gear, as the camera view was often obscured on tows where bottom gear was used (Middleton & Banks 2008). The SLED Working Group was disbanded in early 2010.

Table 4.9: Maximum allowable level of fisheries-related mortality (MALFiRM) or fisheries-related mortality limit (FRML) from 1991 to 2015. Note, however, that direct comparisons among years of the limits in Table 4.9 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%	

4.4.2.2 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 lions interactions on observed tows in a given fishery. This enabled a robust estimation of the total number of New Zealand sea lions killed. However, following the introduction of SLEDs, the number of New Zealand sea lions interacting with trawls but exiting via the SLED was unobservable, so interaction rate is instead estimated statistically. Subsequently, a management setting meant to approximate the interaction rate, i.e., the 'Strike Rate' is set by MPI (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 is then used to set an effort limit on the operation of the fishery, to ensure that sea lion fisheries mortalities remain below the FRML.

The 'Discount Rate' is a management setting that approximates SLED efficacy, i.e., the proportion of sea lion interactions in which the sea lion exits the SLED and survives. The current management regime for the SQU 6T fishery provides that the discount rate will be applied to all tows in which an approved Mark 13/3 SLED is used and relevant requirements of the Operational Plan met (e.g., notification and reporting requirements). Discount rates applied between 2003–04 and 2017–18 are shown in Table 4.9.

The SLED Discount Rate is 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 may be set deliberately lower than the actual estimated SLED efficacy rate, reflecting cautious management in the presence of uncertainty. New research is planned to provide more robust estimates of SLED efficacy (see Section 4.4.3.1, Cryptic mortality, below).

4.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Consistent with terminology used in the SEFRA methodology (Chapter 3), MPI 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 reflects policy decisions.

For Auckland Islands sea lions, the estimation of the PST derives 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 in Roberts & Doonan (2016). The model also supported a quantitative risk assessment to estimate the effects of non-fishery threats (Section 4.4.3.4, 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 the 2014-15 and 2015-16 mark-recapture data (Roberts in prep). These outputs were then used to update the Operational Plan for the SQU 6T fishery in late 2017. Applying the chosen population reference outcome -- i.e. '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 model yielded a corresponding PST of 46 annual fishery-related deaths for the Auckland Island population. Deducting an estimated 7 annual mortalities for the SCI 6A fishery, and one for estimated mortality in other trawl fisheries adjacent to the Auckland Islands, results in a FRML of 38 for the SQU 6T fishery (Table 4.9).

Separately, a simpler PBR approach has been used to identify the level of fishery interactions that would cause adverse effects on the Campbell Island sub-population (Roberts et al. 2014b).

4.4.3.1 CRYPTIC MORTALITY

SLEDs are effective in allowing most New Zealand sea lions to exit a trawl 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 exit 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 five potential fates 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 net but dies from head injuries sustained during interaction with the SLED (‘mild traumatic brain injury’, or MTBI);
v. exits the SLED but is at the limit of its breath hold ability and drowns before reaching the surface (‘post-escape drowning’).

Collectively, points iii–v constitute cryptic mortality. The following section describes research already undertaken to estimate various components of cryptic mortality; additional research is ongoing.

4.4.3.1.1 POST-ESCAPE DROWNING

Between 1999–2000 and 2002–03, an experimental approach was taken to assess the potential for post-escape drowning, 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 the low number of captures filmed and the inability to assess longer-term survival, this work was judged to be inconclusive (Roe 2010).

4.4.3.1.2 MILD TRAUMATIC BRAIN INJURY

Necropsies were conducted on animals recovered from the cover net trials and on those incidentally caught and recovered from vessels operating in the SQU 6T, SQU 1T and SBW 6I fisheries. All of the 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, New Zealand 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). Any head trauma, most likely due to impacts with the SLED grid, could not be ruled out as a potential contributing factor (Roe & Meynier 2010, Roe 2010). In order to quantify the likelihood of a New Zealand sea lion experiencing physical trauma sufficient to render the animal insensible (and therefore likely to drown) after a collision with a SLED grid, a number of factors need to be assessed. These include the likelihood of a head-first impact, the speed of impact, the angle of impact relative to individual grid bars and relative to the grid plane, the location of impact on the grid, head mass, and the risk of brain injury for a given impact speed and head mass. The effect of multiple impacts also needs to be considered. Estimates for each of these factors were obtained from a number of sources, including necropsies (for head mass), video footage of Australian fur seals interacting with Seal Exclusion Devices (SEDs) (for impact speed, location and body orientation) and biomechanical modelling of impacts on the SLED grid (for the risk of brain injury).

In the absence of sufficient video footage of New Zealand sea lions interacting with SLEDs, footage of fur seals (thought to be Australian fur seals) interacting with SEDs in the Tasmanian small pelagic mid-water trawl fishery has been used (Lyle 2011). The SEDs are similar, but not identical, to the New Zealand SLEDs in that both have sloping steel grids to separate the catch from pinnipeds and guide the latter toward an escape hole in the trawl. The angle of slope and the number of sections in the steel grids are variable (either two or three sections, depending on the vessel). Lyle & Willcox (2008) conducted a camera trial between January 2006 and February 2007 to assess the efficacy of the SED and documented 457 interactions for about 170 individual fur seals. Lyle (2011) reanalysed the footage to estimate impact speed, impact location across the SED grid and body orientation at the time of impact. The situation faced by New Zealand sea lions in a squid trawl is different to that faced by the fur seals studied by Lyle and co-workers, but these are closely related otariids of similar size and, in the absence of specific data, Australian fur seals are considered a reasonable proxy to estimate impact speed, impact location and body orientation.

The risk of brain injury was assessed by biomechanical testing and modelling. Tests using an artificial 'head form' (as used in vehicular 'crash test' studies) were used to assess the likelihood of brain injury to New Zealand sea lions colliding with a SLED grid (Ponte et al. 2010, 2011). In an initial trial, the head form (weighing 4.8 kg) was launched at three locations on the SLED grid at a speed of 10 $m.s^{-1}$ (about 20 knots). This was considered a 'worst feasible case' collision, representing the combined velocities of a sea lion swimming with a burst speed of 8 $m.s^{-1}$ (after Ray 1963, Fish 2008) and a net being towed at 2 $m.s^{-1}$ (about 4 knots). A 'head injury criterion' (HIC, a predictor of the risk of brain injury), was calculated based on criteria validated against human-vehicle impact studies and translated into the probability of 'mild traumatic brain injury' (MTBI) for a given collision, taking into account differences between human and sea lion head and brain masses. MTBI is assumed to have the potential to lead to insensibility or disorientation and subsequent death through drowning for a New Zealand sea lion experiencing such an injury at depth. Ponte et al. (2010) calculated that a collision at the stiffest part of the SLED grid at this highest feasible speed had a very high risk of MTBI, especially for smaller sea lions (female and small, immature males). This provides an upper bound for the assessment of risk but Ponte et al. (2010) also imputed risk at speeds below the maximum tested (10 $m.s^{-1}$).

In a follow-up study, after a research advisory group meeting with other experts, Ponte et al. (2011) tested a wider variety of impact locations on the grid and various angles of impact relative to the bars and to the plane of the grid and combined these to produce a HIC 'map' for a SLED grid. This HIC map can be used to estimate the risk of MTBI for a collision by a sea lion at any given speed, location, and orientation used to model the risk of MTBI.

The data collected from the footage of Australian fur seal SED interactions (Lyle 2011) and the biomechanical modelling (Ponte et al. 2010, 2011), were combined in a simulation-based probabilistic model to estimate the risk of a sea lion suffering a mild traumatic brain injury when striking a SLED grid (Abraham 2011). The simulation involved selecting an impact location on the SLED grid (from the fur seal data), selecting a head mass (from New Zealand sea lion necropsy data) and an impact speed (from the fur seal data), calculating the head impact criterion (from the HIC map), scaling the HIC to the head mass and impact speed and calculating the expected probability of mild traumatic brain injury, MTBI. Both 45° and 90° impacts were considered, with the former reflecting the angle of a grid when deployed, adopted as the base case. The head masses used may be at the lower end of the range of head masses for New Zealand sea lions, due to the possible bias in those that were caught and necropsied. Impact speeds were drawn from the distribution of speeds observed for fur seals colliding with SEDs (2–6 $m.s^{-1}$) and these are broadly consistent with the combined tow speed and observed swimming speeds of New Zealand sea lions in the wild (Crocker et al. 2001). Different scaling of HIC values was assessed to gauge sensitivity.

For the base case, the simulation results indicated there was a 3.3% chance of a single head-first collision resulting in MTBI with a 95 percentile of 15.7% risk of MTBI (Abraham 2011). Sensitivities modulating single parameters resulted in up to 6.2% probability of a single collision resulting in MTBI. One sensitivity trial involving changes in multiple parameters resulted in a 10.9% probability of MTBI. This scenario considered impact speeds 20% above those measured for fur seals, multiple collisions with the grid, and the least favourable values of scaling exponents used in scaling the test HIC values and calculating MTBI from the HIC (Abraham 2011). These results are probabilities of MTBI resulting from a single head-first collision but, because each individual can have multiple interactions with the grid while in a trawl, and some of these will not be head-first, the risk of a head collision that results in MTBI for an individual animal over the course of a tow will then be higher than the risk associated with a single interaction sequence. Using Australian observations, Abraham (2011) estimated the number of head-first collisions per interaction as 0.74, leading to an estimated probability of MTBI for a New Zealand sea lion interacting with a trawl of 2.7%. Single parameter sensitivity runs increased this to up to 4.6% and the multiple parameter sensitivity using the scenario described above increased it to 8.2% (Abraham 2011). Assuming synergistic interaction between successive head-first strikes (each collision carrying five times more risk than previous ones) did not appreciably increase the overall risk because few fur seals had multiple head-first collisions. These results indicate that rates of death by MTBI for New Zealand sea lions interacting with the SLED grid is probably low, although some remaining areas of uncertainty were identified (see below).

4.4.3.1.3 BODY NON-RETENTION

Due to obstructed visibility during camera trials in the New Zealand squid fishery, potential rates of body non-retention in SLEDs have not been directly estimated.

From first principles and considering SLED design (Figure 4.7) it seems unlikely that body non-retention rates are high, because:

i) the escape hatch of SLEDs employed in New Zealand fisheries is at the top of the net, while drowned pinnipeds are observed to be negatively buoyant;
ii) forward-facing hoods are designed to allow exit only for actively swimming animals;
iii) hood floats are designed to close the escape hatch in the event that the net becomes inverted (turns upside down), and fishers have a strong incentive to avoid tangling their gear in this way.

Preliminary results of SLED monitoring trials in overseas jurisdictions appear to support the conclusion that drowned pinnipeds are unlikely to be lost, and thereby not counted among observed captures, in trawls employing SLEDs. Collaboration with overseas researchers is ongoing to optimize the use of all available data. Developing or accessing improved means of estimating body non-retention is a high priority.

4.4.3.2 PBR ASSESSMENT FOR CAMPBELL ISLAND POPULATION

Following an unprecedented number of incidental captures of New Zealand sea lions in the Campbell Rise Southern blue whiting fishery (SBW 6I) in 2013, the Deepwater Group requested an expedited audit to assess whether or not the fishery was still in conformance with the Marine Stewardship Council Fisheries Standard (i.e., were interactions below the level that would cause adverse effects on sea lion population size). A review was conducted of 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 calculated as:

$$PBR = N_{min} \times {R_{max}}/{2} \times F_R$$

where R_{max} is the population growth rate at very low population size with only natural mortality operating, N_{min} is a 'minimum' estimate of the total population size and F_R is a recovery factor applied to account for uncertainty or biases that may otherwise lead to overestimation of the PBR and so hinder recovery to an optimum sustainable population (OSP) level. The value of F_R may also be adjusted to meet different population management objectives.

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 sub-populations are likely to constitute demographically independent populations and so, according to the latest guidelines on PBR assessment, may be assessed as separate stocks (Moore & Merrick 2011). Therefore the recovery factor (F_R) of 0.5 was used for stocks of a threatened species with unknown (or not declining) population trajectory. The latest PBR guidance literature recommends a more conservative F_R of 0.1 for stocks of an endangered species and is the lower limit that might be considered for declining populations of a threatened species (Roberts et al. 2014b).

Previous 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 4.8). Running means of capture levels (3- and 5-year) were also calculated for comparison with PBR estimates. For an 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 a 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).

4.4.3.3 INTEGRATED BAYESIAN MANAGEMENT PROCEDURE EVALUATION MODEL

Note that from 2016, on the advice of the AEWG, the 'BFG model' and its precursors described in this section are no longer used to inform MPI management, having been replaced by the integrated Bayesian multi-threat risk model described in Roberts & Doonan (2016). The description in this section is only retained for historical completeness.

From 2000 to 2011, an integrated Bayesian management procedure evaluation model having both population and fishery components was used to assess the likely performance of a variety of management control rules, each of which could be used to determine the FRML for a given SQU 6T season (Breen et al. 2003, 2016, Breen & Kim 2006a, 2006b). The model underwent several iterations. An early version, developed in 2000–01, was a relatively simple, deterministic, partially age-structured population model with density-dependence applied to pup production (Breen et al. 2003). An updated version called the Breen-Kim model was built in 2003 to render it fully age-structured and to incorporate various datasets supplied by DOC (Breen & Kim 2006a, 2006b). This model was further revised in 2007–08 to incorporate the latest New Zealand sea lion population data and to address various model uncertainties and called the BFG model (after its authors, Breen et al. 2016). In 2009, the model was again updated to incorporate the low New Zealand sea lion pup counts observed in 2008–09 (and thus better reflect the observed variability in pup survival and pupping rates), as well as incidental captures in fisheries other than SQU 6T. The BFG model was re-run in 2011 using the same underlying data and structure as in 2009 to evaluate the effect of different model assumptions about the survival of New Zealand sea lions that exit trawl nets via SLEDs (see below). Additional details can be found in Breen et al. (2016).

The BFG model incorporated various population dynamics observations (tag-resighting observations, pup births and mortality, age at maturity) as well as incidental captures and catch-at-age data from the SQU 6T trawl fishery. The model was projected into the future by applying the observed dynamics and a virtual fishery model that is managed in roughly the same way as the real SQU 6T fishery. A large number of projections were run and used to assess the likely performance of a wide range of different management control rules against defined performance criteria.

4.4.3.4 COMPREHENSIVE QUANTITATIVE RISK ASSESSMENT

In 2016 a quantitative risk assessment estimating the potential impacts of both fisheries and non-fishery threats to the New Zealand sea lion (*Phocarctos hookeri*) was undertaken to inform the development of a Threat Management Plan for the species (Roberts & Doonan 2016).

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 June 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, in order 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 populations, 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 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 was 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 in order 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 4.8, and for the Otago Peninsula population in Figure 4.9. The black line in Figures 4.8 and 4.9 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., *Klebsiella pneumoniae* (a bacterial disease), hookworm, wallows[7]) 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 as 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).

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 4.9). 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 4.10). 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 4.11, Roberts & Doonan 2016).

For the Otago Peninsula population, similar effects were estimated with the alleviation of any of the threats that

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

passed through triage: commercial set-net fishery-related mortality; direct human mortality; pollution-related 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 4.12, Roberts & Doonan 2016).

For the Auckland Islands population (the largest breeding site for the species), model outputs suggest that if demographic rates used to simulate forward population trajectories (i.e., sampled from the past 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 (Roberts et al. in prep). 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 new work to examine factors affecting pup survival (PRO2017-08C) will commence in 2018.

The assessment for some of the key threats 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 commercial trawl-related mortality. The separate assessment of climate and fishery effects on food limitation and population growth rate was not included for any population, but has been progressed separately (Roberts et al. in prep). In addition, a lack of demographic observations for the Campbell Island and Stewart Island populations (the second- and third-largest breeding populations, respectively), precluded the development of comprehensive quantitative risk assessments for these populations (Roberts & Doonan 2016).

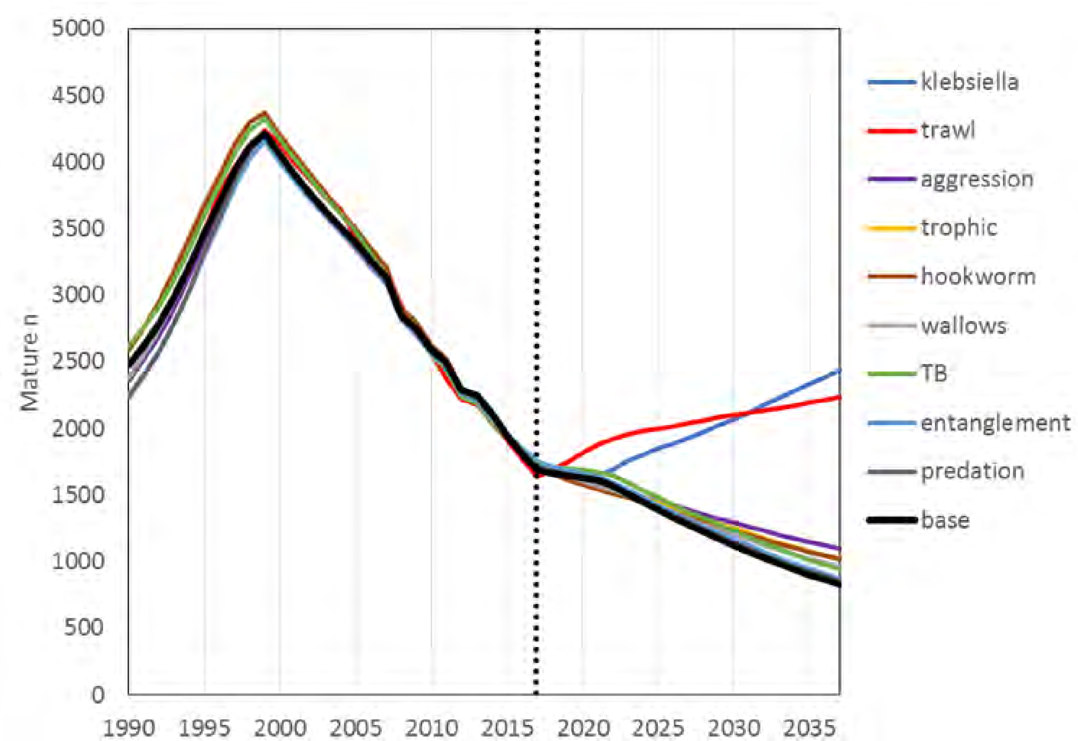


Figure 4.8: Triage projections of model estimated mature n at the Auckland Islands in the period 1990–2037, using upper values of threat mortality. The black dotted line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines.

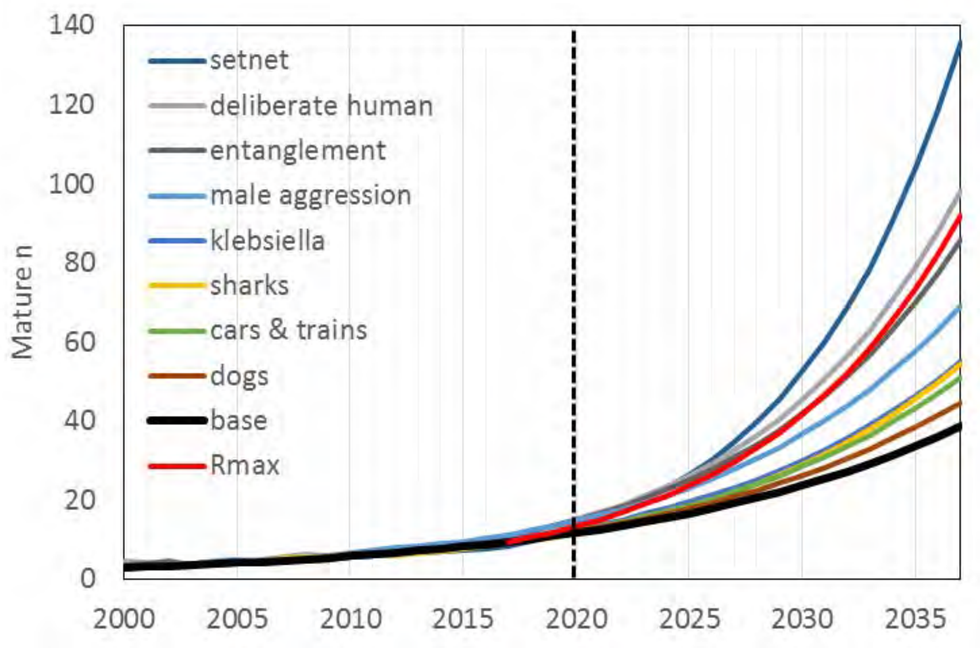


Figure 4.9: Triage projections of model estimated mature n at the Otago Peninsula in the period 1990–2037, using upper values of threat mortality. The black dashed line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal 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) (from Roberts & Doonan 2016).

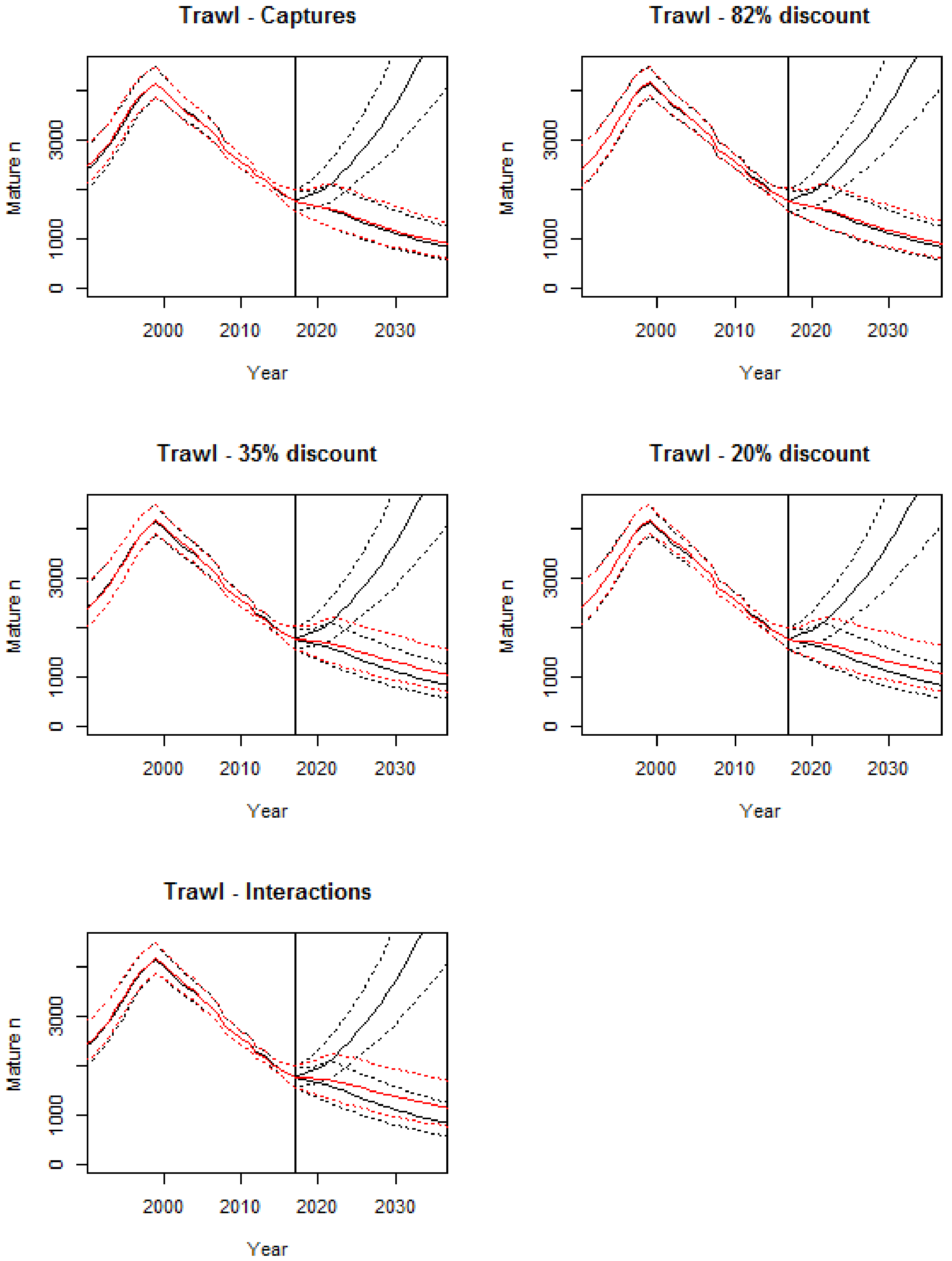


Figure 4.10: Best-estimate projections of mature n at the Auckland Islands in the period 1990–2037 for trawl fishery mortality scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Surv6–14 and 1990–99 estimate of PrP; red lines are with a threat alleviated (from Roberts & Doonan 2016).

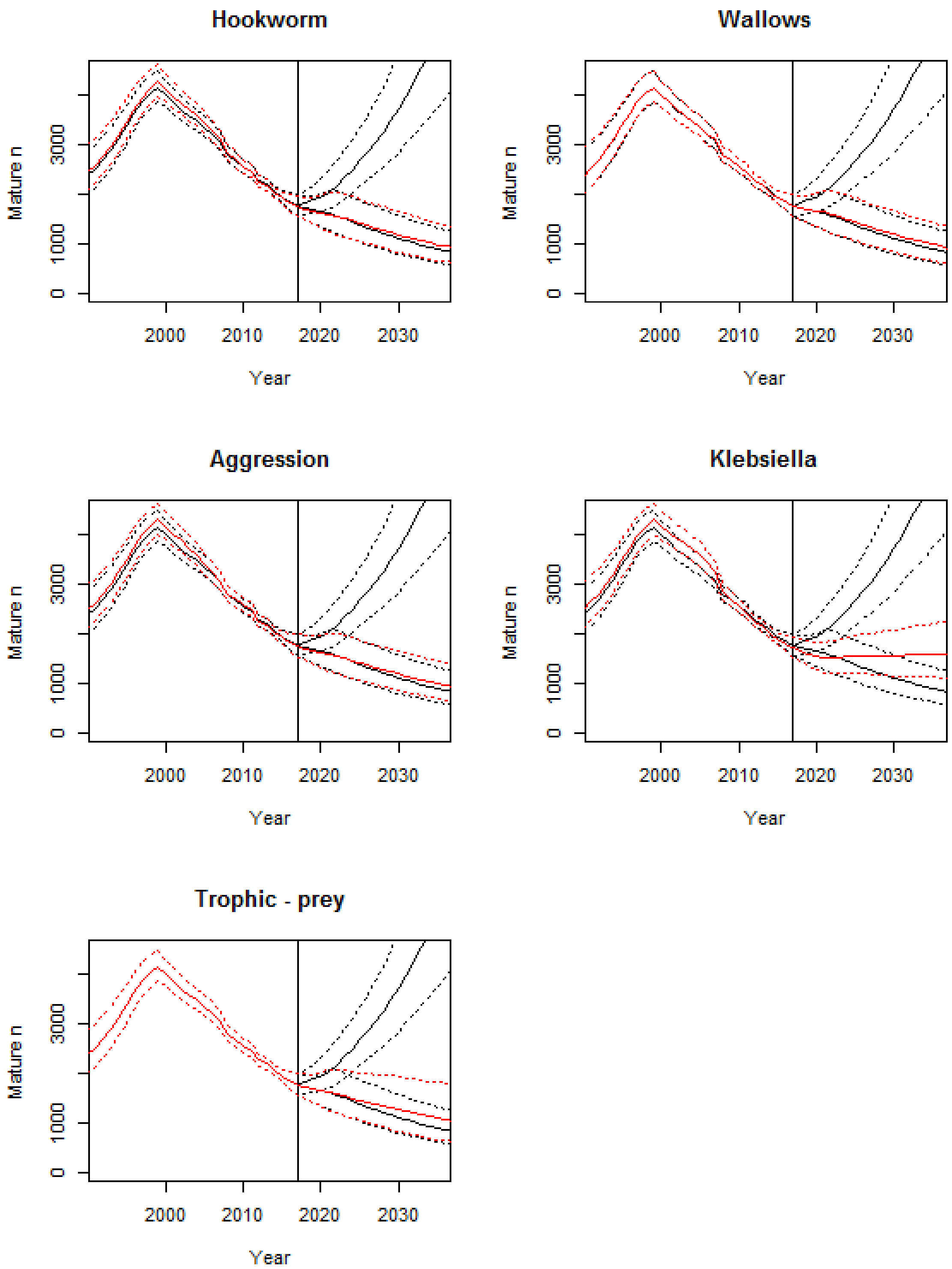


Figure 4.11: Best-estimate projections of mature n at the Auckland Islands in the period 1990–2037 for all other threat scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Surv6–14 and 1990–99 estimate of PrP; red lines are with a threat alleviated (from Roberts & Doonan 2016).

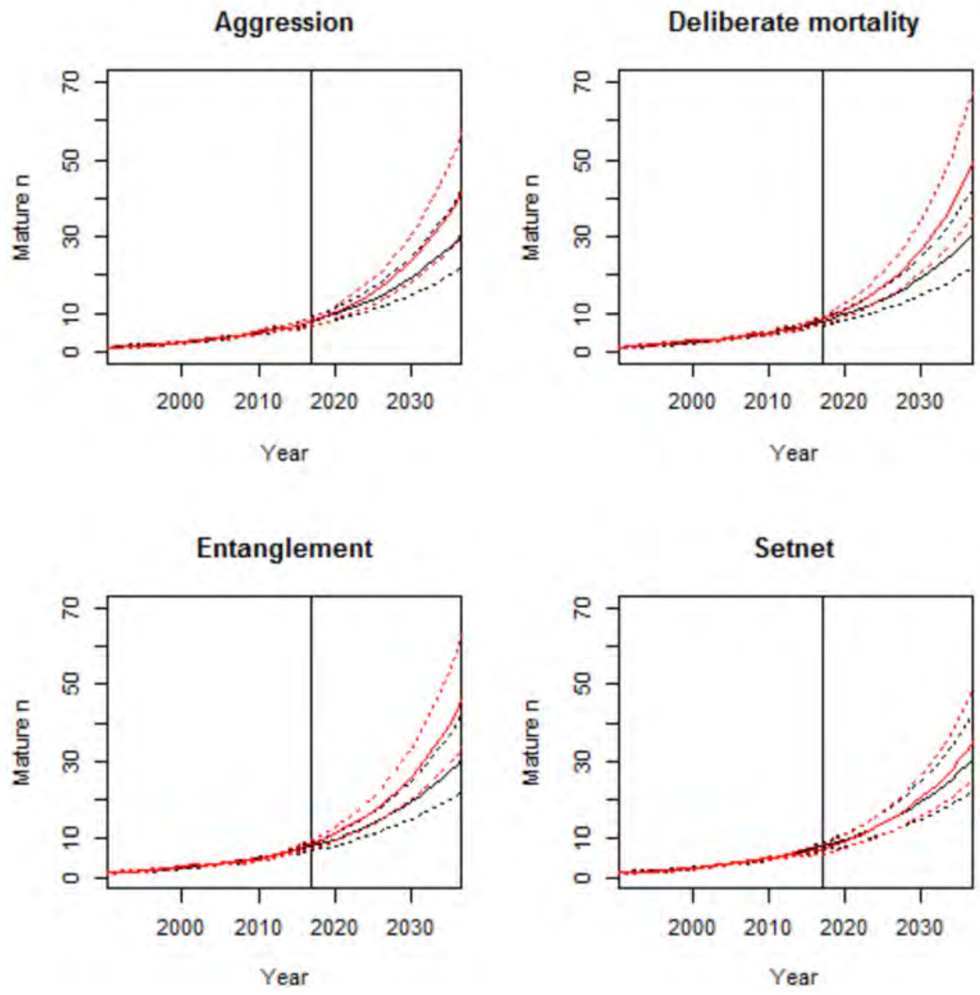


Figure 4.12: Best-estimate projections of mature n at the Otago Peninsula in the period 1990–2037 for all threat scenarios (from Roberts & Doonan 2016). Black lines are with all threats (base run); red lines are with the threat alleviated.

4.4.3.5 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017 the first iteration of a New Zealand Marine Mammal Risk Assessment (NZMMRA) was completed (Abraham et al. 2017), applying an adaptation of the Spatially Explicit Fisheries Risk Assessment (SEFRA) method described in Chapter 3.

The risk assessment outputs suggest that sea lions are the twelfth-highest at-risk species of marine mammal from New Zealand commercial fisheries. Fisheries risk to sea lions is attributable to a range of trawl fisheries. Estimated annual potential fishery-related deaths for sea lions by fishery group are shown in Figure 4.13.

The estimated cumulative fisheries risk score for sea lions is well estimated and ranges from approximately 0.1 to 0.2 (Figure 4.14), consistent with species-specific population modelling results suggesting that direct fisheries-related mortality is unlikely to be a primary cause of sea lion deaths (Roberts and Doonan 2016).

Because sea lions have been the subject of considerable directed data collection and research over many years, and sea lion captures occur primarily in well-observed deepwater or pelagic trawl fisheries, the results of the multi-species NZMMRA in its current form are useful to supplement but not replace the conclusions of more focused population research such as presented by Roberts and Doonan (2016) (see Section 4.4.3.4, Comprehensive quantitative risk assessment, above). An independent external review of the SEFRA method (Lonergan et al. 2017) noted that the reliability and specific applicability of the current 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 NZMMRA 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 sea lions. 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 is currently in progress for sea lions around the Auckland Islands (PRO2017-10), including re-application of the SEFRA method in a species-specific context. Because sea lions show sex-specific movement patterns and foraging behaviours, and because male and female deaths are likely to have very different implications for the population response of harem-breeding mammals, future analyses will likely by necessity be sex-specific, or limited to females only (as in the population modelling of Roberts & Doonan 2016).

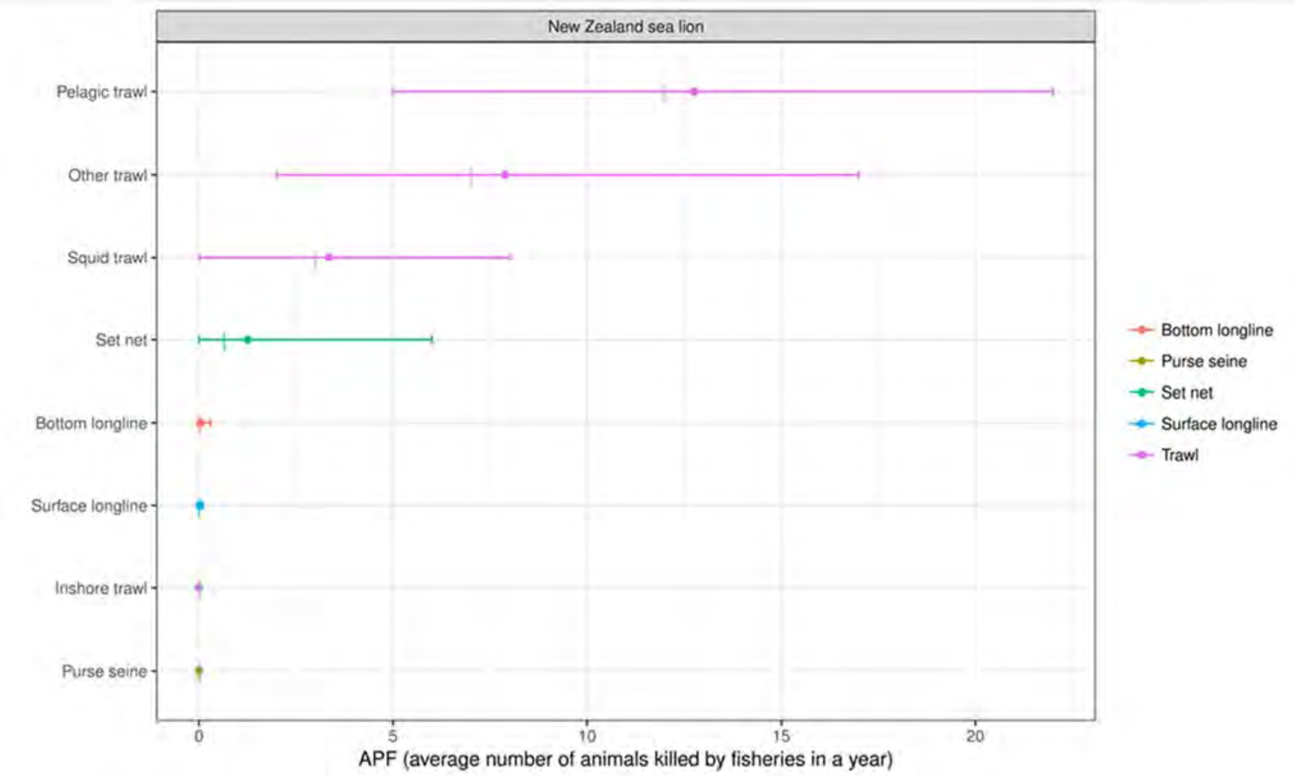


Figure 4.13: Estimated annual potential fishery-related deaths of sea lions by fishery group, as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (Abraham et al. 2017).

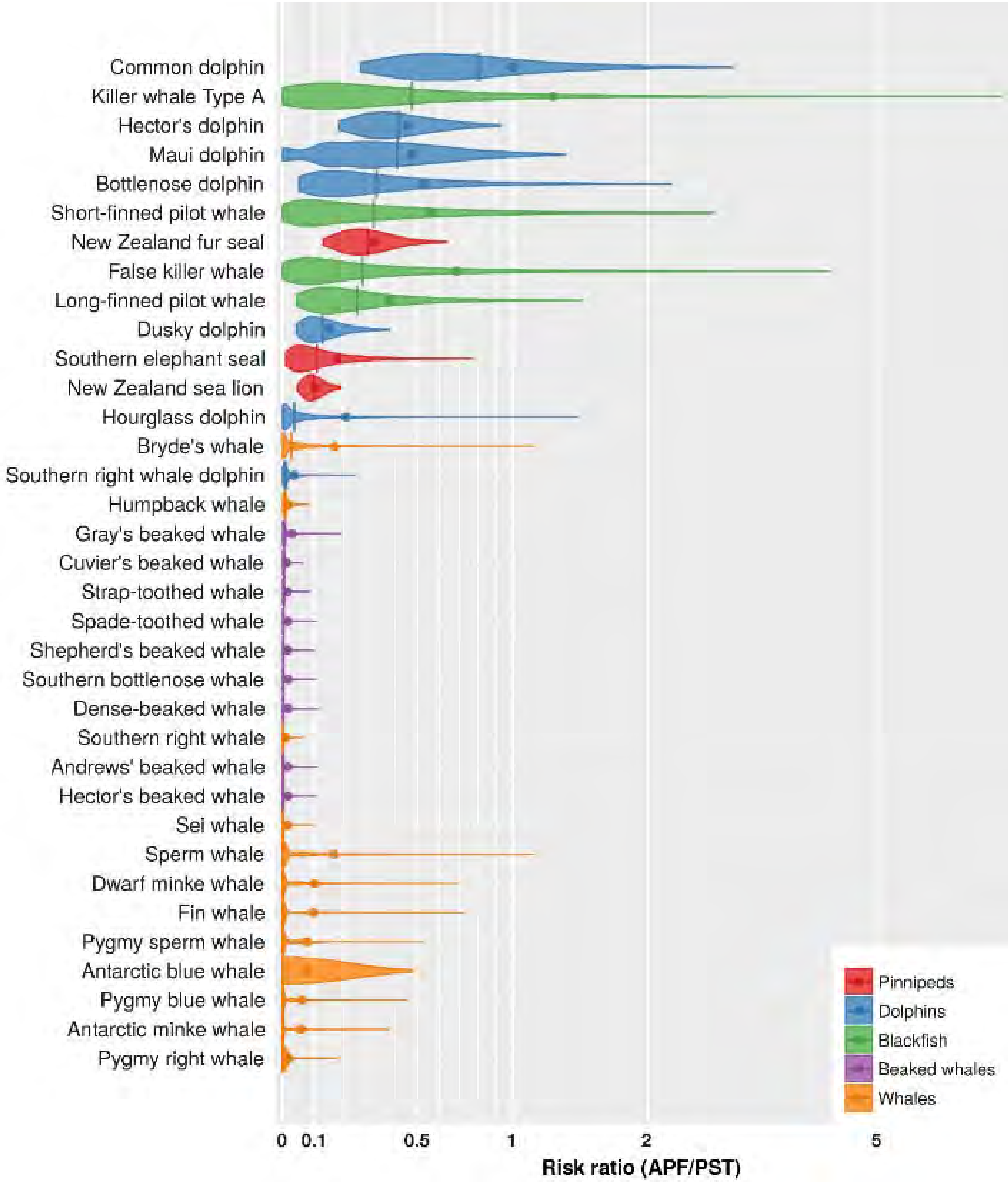


Figure 4.14: Cumulative fishery risk across all fishery groups as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

4.4.3.6 SOURCES OF UNCERTAINTY

There are several sources of uncertainty in modelling the effects of fisheries on New Zealand sea lions at the Auckland Islands, including uncertainty regarding the modelling of sea lion interaction rate (Thompson et al. 2013, Abraham & Berkenbusch 2017), uncertainty regarding inputs and structural assumptions affecting the demographic and multi-threat risk assessment model (Roberts & Doonan 2016), uncertainty regarding biomechanical modelling to estimate mild traumatic brain injury in SLEDs (Ponte et al. 2010, 2011, Abraham 2011, Lyle 2011), and uncertainty regarding other critical rates affecting cryptic mortality.

Population projections in Roberts & Doonan (2016) assumed future demographic rates based on historically observed rates from the past 20 years, which includes the period of steepest population decline and not the preceding period of population growth. It is likely that changes in demographic rates reflect changing environmental conditions (Roberts et al. in prep). 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.

The Roberts & Doonan (2016) model makes 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. These responses have become more apparent as the population has decreased, suggesting that changes in carrying capacity may have occurred. The underlying causes of the apparent change is unknown; and it is unknown whether similar changes can be expected in future, and on what time scales. Simulations may be useful to investigate the potential effect of environmental regime shifts affecting demographic rates on a cyclical or stochastic basis on a range of time scales.

In addition to sources of uncertainty in relation to the population model, there are other sources of uncertainty relevant to the management of fisheries interactions. The estimated interaction rate is highly variable and increasingly unbounded as new data are added in recent years (Table 4.5; see also Thompson et al. 2013, Abraham & Berkenbusch 2017). Alternative means of estimating interaction rate, and/or relying on direct estimation of cryptic mortality as a function of observed captures without reference to interaction rate, will be developed in future.

There are a number of possible sources of uncertainty regarding the biomechanical modelling to estimate mild traumatic brain injury (Ponte et al. 2010, 2011, Abraham 2011, Lyle 2011). The use of linear acceleration, as opposed to rotational (angular) acceleration, may underestimate the risk of MTBI, although this was thought to be accounted for at least in part by sensitivity analysis of the scaling of HIC values. The testing used an artificial 'head form' based on human anatomy, so the effect of New Zealand sea lion scalp thickness and skull morphology is unknown, although differences in head and brain masses are accounted for. Potential effects of differences in the angle of the head on impact (relative to the neck) were not tested. Impact speeds, locations and orientations of New Zealand sea lions may differ from those of Australian fur seals, although the fur seal data were considered to be a reasonable proxy by a Research Advisory Group. The head mass values used may be lower than average for New Zealand sea lions; this would mean risk is likely to be overestimated.

As described above, other potential sea lion fates resulting in cryptic mortality (post-escape drowning, body non-retention) are inherently difficult to estimate and remain uncertain. New research to improve estimates of cryptic mortality is a priority.

4.4.3.7 POTENTIAL INDIRECT THREATS

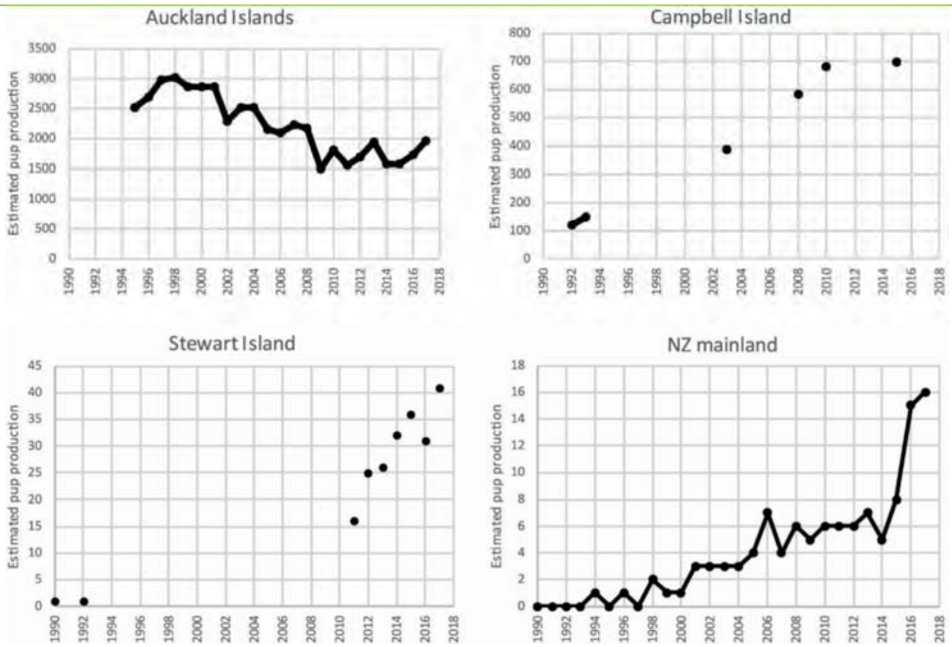
It is possible that indirect fisheries effects may have population-level consequences for New Zealand sea lions. Such indirect effects may include competition for food resources between various fisheries and New Zealand sea lions (Robertson & Chilvers 2011; Roberts et al. in prep). In order to determine 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). A recent study focused on energy and amino acid content of prey, determined that the selected prey species contained all essential amino acids and were of low to medium energy levels (Meynier 2010). 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 the amount of prey consumed by New Zealand sea lions at 17 871 t (95% c.i.: 17 738–18 000 t) per year. This is equivalent to ~30% of the tonnage of arrow squid, and ~15% of the hoki harvested annually by the fisheries in the subantarctic between 2000 and 2006 (Meynier 2010). 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). The effects of fishing on sea lion prey species are likely to be complicated by food web interactions. Multi-species models may help to assess the extent to which resource competition can impact on sea lion populations. In addition, multi-species models may provide a means for simultaneously assessing multiple drivers of sea lion population change (Robertson & Chilvers 2011), which may be a more effective approach than focusing on single factor explanations for the recent observed decline in New Zealand sea lions (Bowen 2012).

New research investigating the abundance and distribution of sea lion prey species, including via a dedicated trawl survey, suggests that Auckland Islands sea lions have endured a protracted period of nutritional stress, and during unfavourable periods this population may have been limited by the availability of key prey (Roberts et al. in prep). However results are inconclusive pending a more thorough understanding of sea lion diet and foraging behaviour under different environmental conditions.

4.5 INDICATORS AND TRENDS

Population size	11 755 New Zealand sea lions including pups (immediately after pupping) (Roberts & Doonan 2016). It is estimated that there were: 1965 pups born at the Auckland Islands in 2017; 696 pups born at Campbell Island in 2015 (Childerhouse et al. 2015a); 41 pups born at Stewart Island in 2017; and 16 pups born on the Otago Peninsula and Catlins coast in 2017.[8]
Population trend	Estimated annual pup production at the Auckland Islands, Campbell Island, Stewart Island, and New Zealand mainland is shown below. Reproduced from DOC & MPI (2017).

[8] New Zealand Sea Lion Trust, Otago Sea Lion Family Trees, https://www.sealiontrust.org.nz/otago-sea-lion-family-tree.

	
Threat status	New Zealand: Nationally Critical, Criterion C[9], Range Restricted[10], in 2013[11] IUCN: Endangered, A4bd[12], in 2015[13]
Number of captures	12 estimated captures (95% c.i.: 8–17) in trawl fisheries in 2014–15[15] 4 observed captures in trawl fisheries in 2015–16[15]

[9] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of sub-populations) 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).

[10] 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 sub-populations (Townsend et al. 2008).

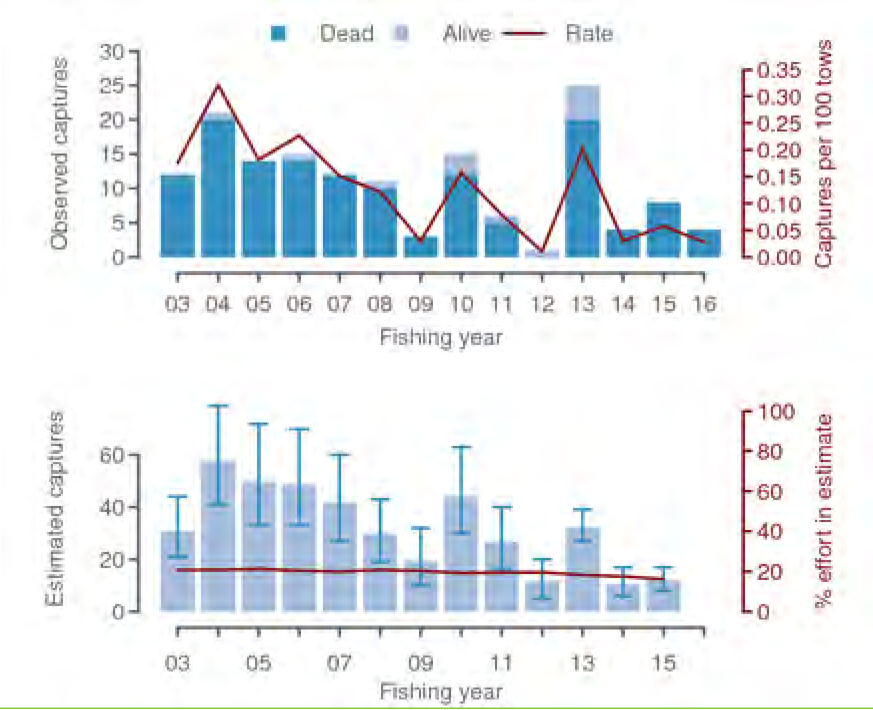
[11] Baker et al. (2016).

[12] 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).

[13] Chilvers (2015). For more information, see: http://data.dragonfly.co.nz/psc.

[16] Abraham et al. (2017).

Trends in observed and estimated captures	



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5 NEW ZEALAND FUR SEAL (*ARCTOCEPHALUS FORSTERI*)

Scope of chapter	This chapter briefly 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	All of the New Zealand EEZ and Territorial Sea, but primarily in coastal environments extending to the continental slope.
Focal localities	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, plus offshore of Bay of Plenty-East Cape. 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, plus offshore of Bay of Plenty-East Cape. Interactions also occur off the west coast of the North Island.
Key issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries; improved understanding of population structure and trend on a regional basis; improved understanding of spatio-temporal distributions affecting interaction rates with 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)	PRO2012-02 *Assess the risk posed to marine mammal populations from New Zealand fisheries*; PRO2012-02 *Assessment of the risk to marine mammal populations from New Zealand commercial* fisheries; PRO2012-02 *Assessment of the risk to marine mammal populations from New Zealand commercial fisheries.*
NZ 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

Note: This chapter has been updated for the 2017 AEBAR with recent capture estimation and risk assessment results.

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 Primary Industries (formerly the Minister of Fisheries), 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*[1] (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.

Under the National Deepwater Plan, the objective most relevant for management of New Zealand fur seals is Management Objective 2.5: *Manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened and protected species*.

Specific objectives for the management of New Zealand fur seal bycatch 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 hoki (HOK), southern blue whiting (SBW), hake (HAK) and jack mackerel (JMA). The HOK chapter of the National Deepwater Plan (completed in 2010) includes Operational Objective (OO) 2.11: *Ensure that incidental marine mammal captures in the hoki fishery are avoided and minimised to acceptable levels (which may include standards) by 2012*. The SBW chapter (2011) includes OO2.3: *Ensure that incidental New Zealand fur seal mortalities, in the southern blue whiting fishery at the Bounty Islands (SBW6B), do not impact the long term viability of the fur seal population and captures are minimised through good operational practices*. The HAK plan (active from 2013–14) includes OO2.4: *Ensure that incidental marine mortalities in hake fisheries are mitigated and minimised*. The JMA plan (active from 2013–14) includes OO2.2: *Ensure that incidental marine mammal captures, particularly common dolphins, do not impact the long term viability of the population and captures are minimised through good operational practices*.

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.*' This comprises four components: Avoid, remedy, or mitigate the adverse effects of fishing on associated and dependent species, including through maintaining food chain relationships; Minimise unwanted bycatch and maximise survival of incidental catches of protected species in HMS fisheries, using a risk management approach; Increase the level and quality of information available on the capture of protected species; and Recognise the intrinsic values of HMS and their ecosystems, comprising predators, prey, and protected species.

The Environment Objective is the same for all groups of fisheries in the draft National Fisheries Plan for Inshore Finfish, to '*Minimise adverse effects of fishing on the aquatic environment, including on biological diversity*'. The draft National Fisheries Plans for Inshore Shellfish and Freshwater have the same objective, but are unlikely to be relevant to management of fishing-related mortality of New Zealand fur seals.

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).

[1] DOC has confirmed that the Marine Mammal Action Plan for 2005–10 still reflects DOC's priorities for marine mammal conservation.

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 species' range and in population decline almost to the point of extinction (Smith 1989, 2005, 2011, Ling 2002, Lalas 2008). 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 can dive to at least 274 m (for a 5.67 min dive in autumn) and remain 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 fed 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, west coast South Island, Otago Peninsula, Kaikoura, 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 on breeding colonies (Crawley & Wilson 1976), or the reproductive success. Boren (2005) reported a fecundity rate of 62% for a Kaikoura 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 (submitted) 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 Kaikoura, 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. 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), Kaikoura (Boren 2005), Otago (Lalas & Harcourt 1995, Lalas & Murphy 1998, Lalas 2008, Lalas & MacDiarmid, submitted), 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. (The NABIS lineage document as well as layer details and associated metadata are available online: http://www2.nabis.govt.nz/LayerDetails.aspx?layer=Breeding%20colonies%20distribution%20of%20New%20Zealand%20fur%20seal.)

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 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 Kaikoura 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 were unsuccessful in areas of high canopy cover (>80%). Ground surveys were also conducted using a higher resolution Optris PL450™ 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™ 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™ 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. 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.

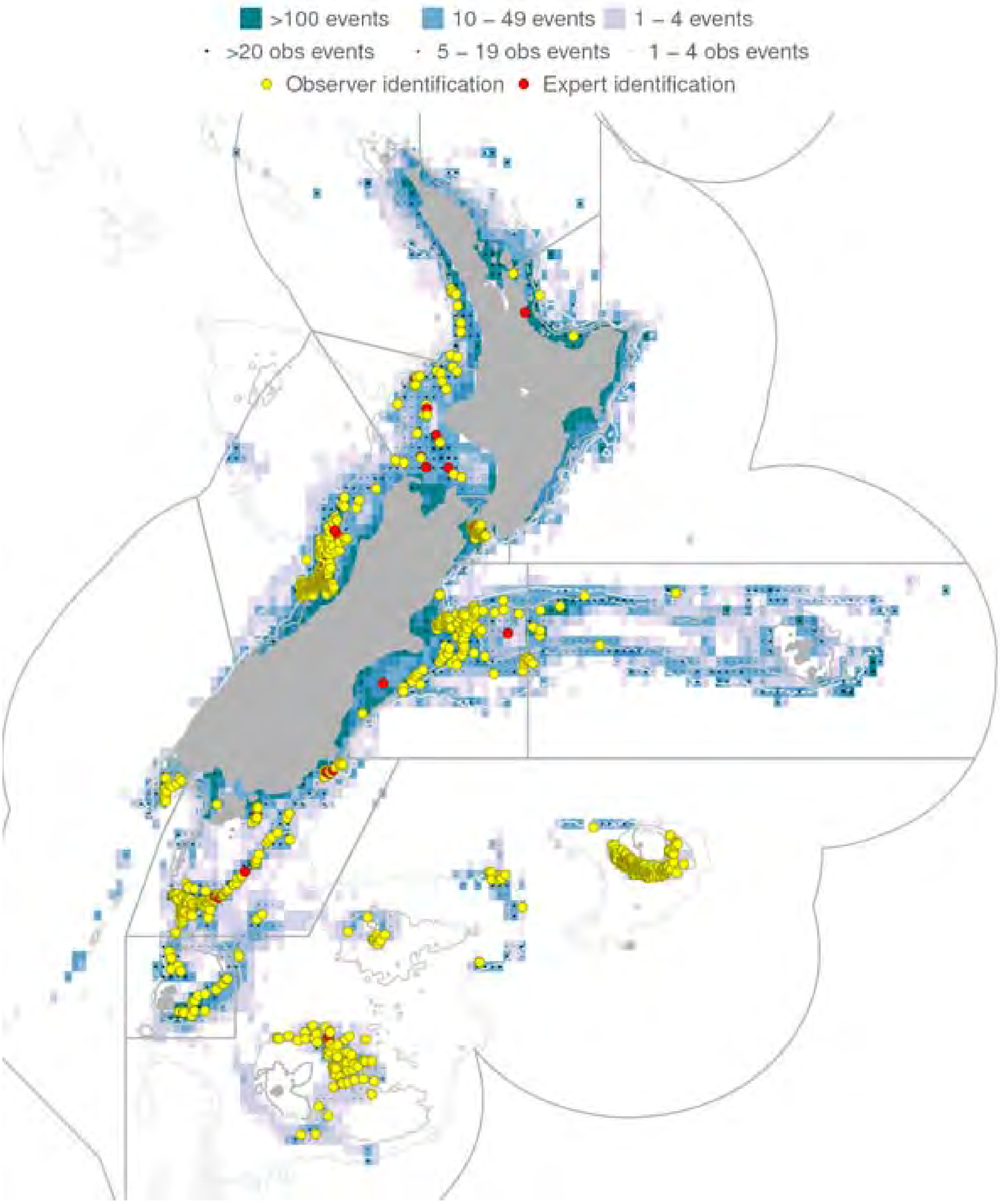


Figure 5.1: Distribution of trawl fishing effort and observed New Zealand fur seal captures, 2002–03 to 2015–16 (for more information see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2017001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell.

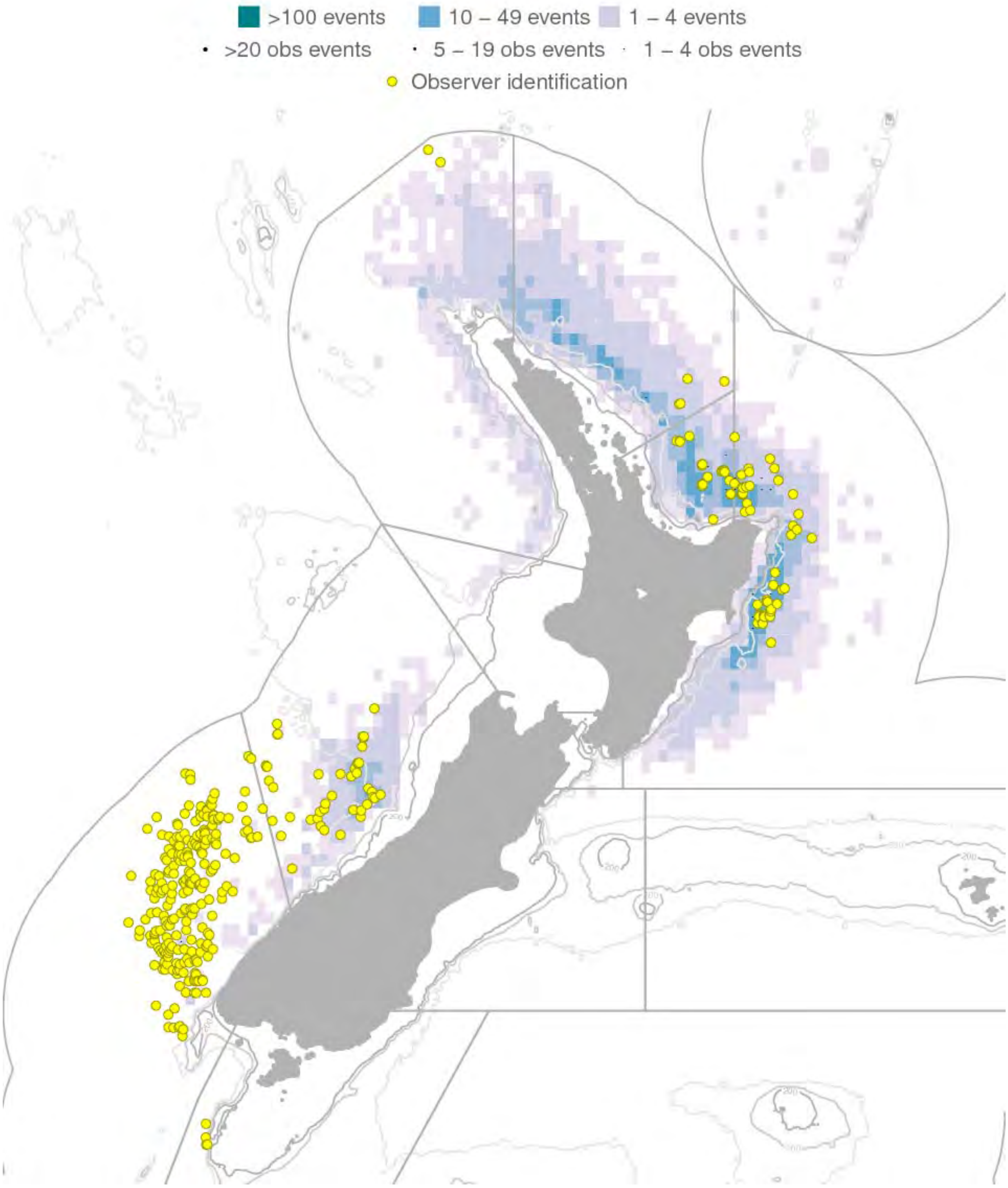


Figure 5.2: Distribution of surface-longline fishing effort and observed New Zealand fur seal captures, 2002–03 to 2015–16 (for more information see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2017001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by red 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, 89.6% of the effort is shown.

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 seal per 100 tows, to give an annual mean total of 486 captures (95% c.i.: 299–876) (Table 5.2, Figure 5.3). Most annual captures are generally observed in Cook Strait. Note these 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 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). Captures associated with recreational fishing activities are poorly known (Abraham et al. 2010a).

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.

<table>
<tr><th>New Zealand fur seals</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th></tr>
<tr><td>Breeding males</td><td>Dispersed at sea or at haulouts</td><td colspan="4">At breeding colony</td><td colspan="7">Dispersed at sea or at haulouts</td></tr>
<tr><td>Breeding females</td><td colspan="2">At sea</td><td colspan="2">At breeding colony</td><td colspan="8">At breeding colony and at-sea foraging and suckling</td></tr>
<tr><td>New pups</td><td colspan="3">At sea</td><td colspan="9">At breeding colony</td></tr>
<tr><td>Non-breeders</td><td colspan="12">Dispersed at sea, at haulouts, or breeding colony periphery</td></tr>
<tr><th>Major fisheries</th><th>Sep</th><th>Oct</th><th>Nov</th><th>Dec</th><th>Jan</th><th>Feb</th><th>Mar</th><th>Apr</th><th>May</th><th>Jun</th><th>Jul</th><th>Aug</th></tr>
<tr><td>Hoki trawl</td><td></td><td colspan="8">Chatham Rise and Stewart-Snares Shelf</td><td colspan="3">Cook Strait, west coast South Island, Puysegur</td></tr>
<tr><td>Squid</td><td colspan="3"></td><td colspan="2">Stewart-Snares Shelf</td><td colspan="6">Auckland Islands and Stewart-Snares Shelf</td><td></td></tr>
<tr><td>Southern blue whiting</td><td colspan="2">Pukaki Rise and Campbell Rise</td><td colspan="9"></td><td>Bounty Islands</td></tr>
<tr><td>Scampi</td><td colspan="12">Mernoo Bank (Chatham Rise) and Auckland Islands</td></tr>
<tr><td>Southern bluefin tuna longline</td><td colspan="6"></td><td colspan="5">South-west South Island</td><td></td></tr>
</table>

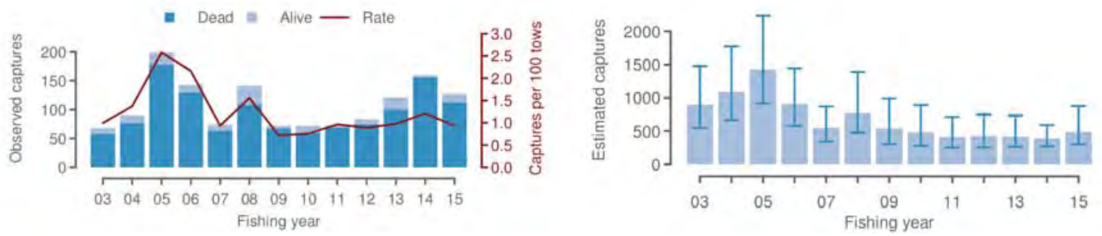
Table 5.2: Fishing effort and observed and estimated New Zealand fur seal captures in trawl and surface-longline fisheries by fishing year in the New Zealand EEZ (Abraham & Berkenbusch 2017, and see MPI data analysis at http://data.dragonfly.co.nz/psc, data version 2016001). For each fishing year, the 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 2017. [Continued on next page]

Fishing year	Fishing effort		Observed captures		Estimated captures	
	All effort	% observed	Number	Rate	Mean	95% c.i.
Trawl fisheries						
2002–03	130 176	5.3	68	0.994	924	562–1 504
2003–04	120 850	5.4	90	1.374	1 120	684–1 839
2004–05	120 461	6.4	199	2.580	1 487	964–2 370
2005–06	109 951	6.0	143	2.160	949	597–1 534
2006–07	103 317	7.7	74	0.933	570	360–898
2007–08	89 536	10.1	142	1.569	795	493–1 406
2008–09	87 550	11.4	72	0.719	564	320–1 026
2009–10	92 889	10.3	72	0.756	495	286–911
2010–11	86 091	8.8	73	0.963	443	262–800
2011–12	84 424	11.1	83	0.889	451	267–804
2012–13	83 832	14.8	121	0.976	438	270–760
2013–14	85 113	15.5	159	1.204	416	291–630
2014–15	78 754	17.6	127	0.915	536	332–969
2015–16	78 040	18.2	109	0.768		

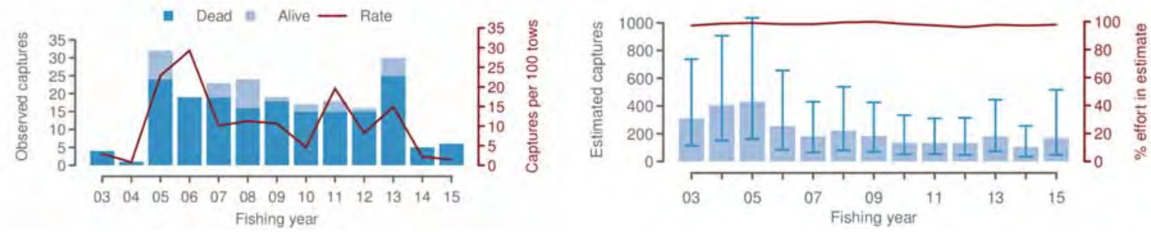
Table 5.2 [Continued]:

Surface-longline fisheries						
2002–03	10 771 038	20.4	56	0.026	369	262–507
2003–04	7 386 339	21.8	40	0.025	148	108–200
2004–05	3 679 965	21.3	20	0.026	68	46–95
2005–06	3 691 809	19.1	12	0.017	51	30–78
2006–07	3 740 012	27.8	10	0.010	29	18–45
2007–08	2 246 689	18.8	10	0.024	41	23–64
2008–09	3 114 733	30.1	22	0.023	54	38–74
2009–10	2 996 544	22.5	19	0.028	83	53–118
2010–11	3 186 899	21.2	17	0.025	67	43–97
2011–12	3 100 277	23.5	40	0.055	146	106–193
2012–13	2 876 932	19.5	21	0.037	109	71–156
2013–14	2 549 764	30.7	57	0.073	176	137–224
2014–15	2 412 336	30.1	37	0.051	116	87–151
2015–16	2 359 891	13.7	3	0.009		

a) EEZ



b) Cook Strait



c) East Coast South Island

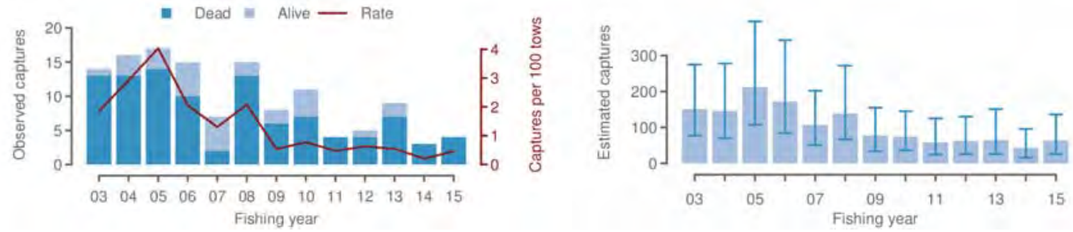
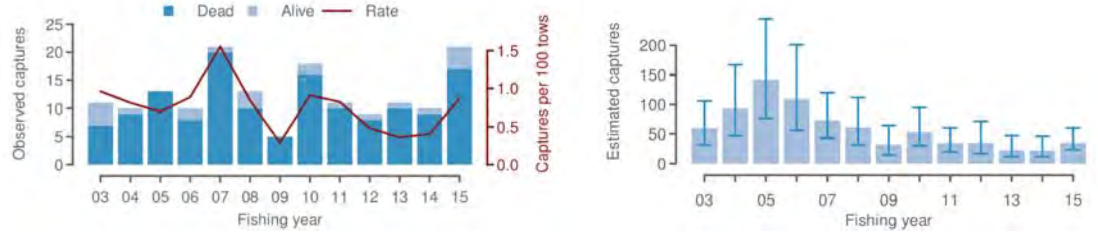
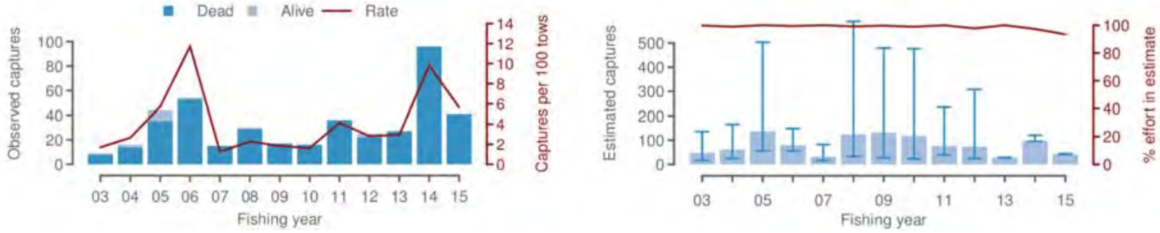


Figure 5.3: Observed captures of New Zealand fur seals (dead and alive) in trawl fisheries, capture rate (captures per hundred tows), and mean number of estimated total captures (with 95% confidence interval) by fishing year for regions with more than 50 observed captures since 2002–03: (a) New Zealand's EEZ; (b) the Cook Strait area; (c) the East Coast South Island area; (d) the Stewart-Snares Shelf area; (e) the subantarctic area; and (f) the West Coast South Island area (Abraham et al. 2017, and see MPI data analysis at http://data.dragonfly.co.nz/psc, data version v2016001). 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 and Thompson (2011). [Continued on next page]

d) Stewart-Snares Shelf



e) Subantarctic area



f) West Coast South Island

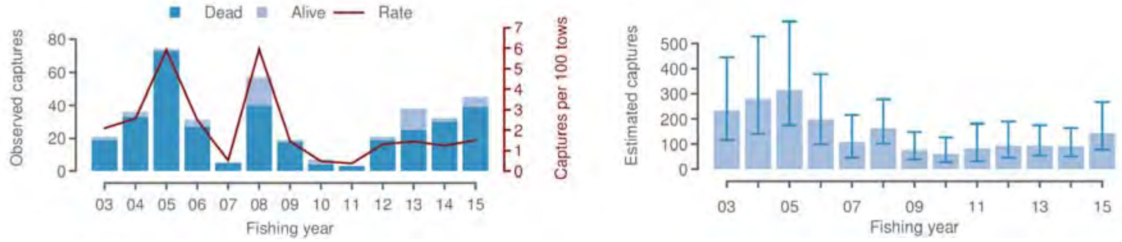


Figure 5.3 [Continued]: Observed captures of New Zealand fur seals (dead and alive) in trawl fisheries, the capture rate (captures per hundred tows), and the mean number of estimated total captures (with 95% confidence interval) by fishing year for regions with more than 50 observed captures since 2002–03: (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 http://data.dragonfly.co.nz/psc, data version v2016001). 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 and Thompson (2011).

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 at most colonies; 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. It 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 >50m
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 sea lions 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 may prove useful. Alternatively, genetics research may also 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 an adaption 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 secondarily to set net fisheries. Estimated annual potential fishery-related deaths for fur seals by fishery group are shown in Error! Reference source not found..

The estimated cumulative fisheries risk score for fur seals ranges from approximately 0.2 to 0.6, consistent with colony observations indicating a general trend of increasing population size in recent years (Figure 5.5). Note that unlike the NZSRA, the NZMMRA does not utilise population monitoring results directly in the risk assessment to constrain total fishery related deaths to be consistent with observed adult survival rates. Introducing this constraint is a priority for future updates of the NZMMRA.

An independent external review of the SEFRA method (Lonergan et al. 2017) noted that the reliability and specific applicability of the current 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 is currently in progress to improve available distribution models for cetaceans (under contract PRO2014-01), for Māui and Hector's dolphins (PRO2017-12) and for Auckland Island sea lions (PRO2017-09); new research applying a similar approach for New Zealand fur seals is planned. 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.

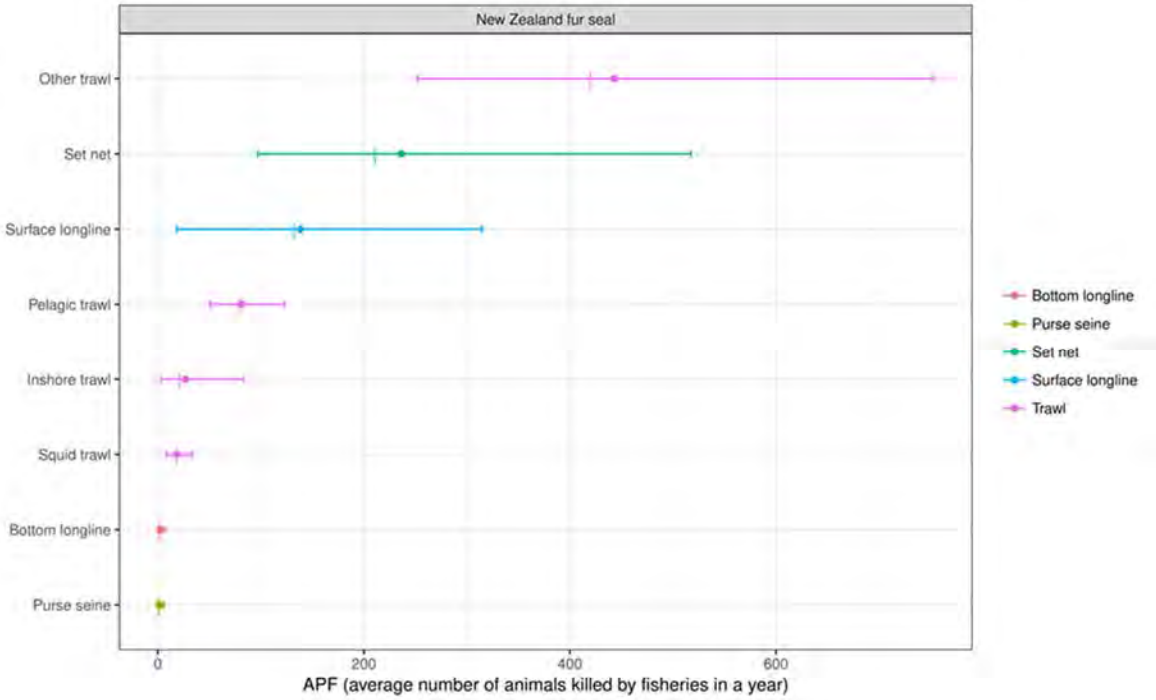


Figure 5.4: Preliminary estimates of annual potential fishery-related deaths of fur seals by fishery group, as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

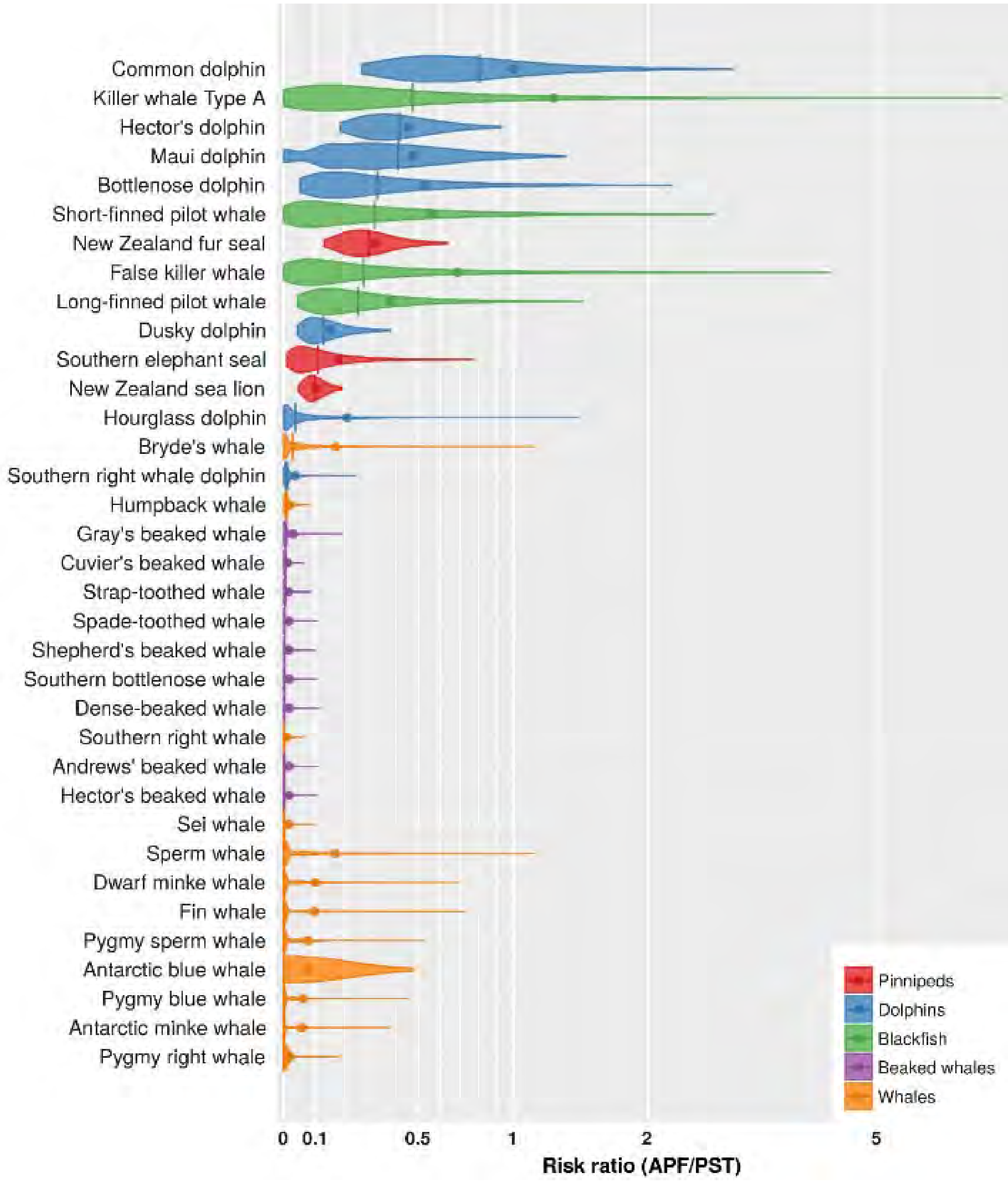


Figure 5.5: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

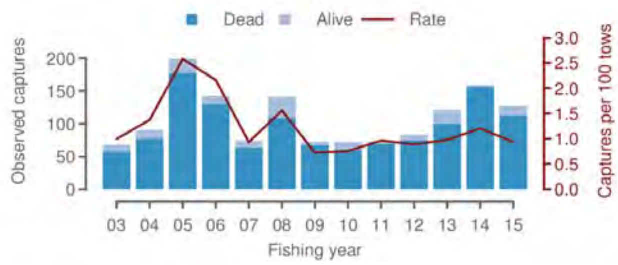
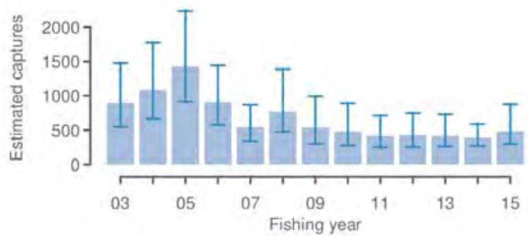
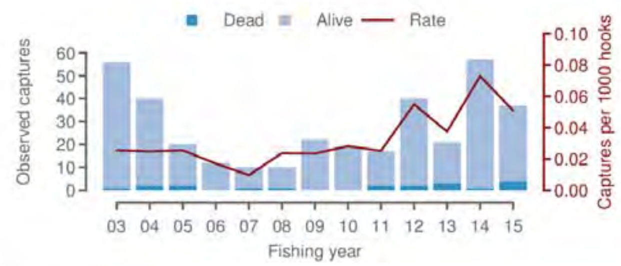
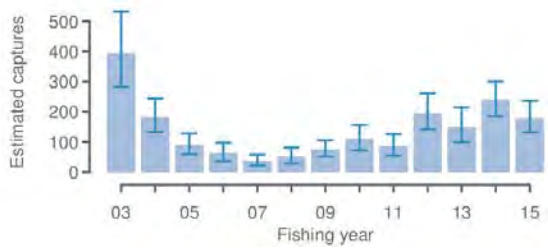
SOURCES OF UNCERTAINTY

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 exist, the size and dynamics of the overall populations 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 Kaikoura. 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	486 estimated captures (95% c.i.: 299–876) in trawl fisheries in 2014–15[6] 179 estimated captures (95% c.i.: 132–237) in surface-longline fisheries in 2014–15[6] 127 observed captures in trawl fisheries in 2014–15[6] 37 observed captures in surface-longline fisheries in 2014–15[6] 949.3 estimated annual potential fatalities (APF) (95% c.i.: 949.3–1 406.5)[7]
Trends in interactions[6]	Trawl fisheries:   Surface-longline fisheries:  

[2] Taylor (1990), Harcourt (2001).
[3] Baker et al. (2016).
[4] Chilvers & Goldsworthy (2015).
[6] For more information, see: http://data.dragonfly.co.nz/psc.
[7] Abraham et al. (2017).

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

Scope of chapter	This chapter briefly describes: the biology of Hector's dolphin (*Cephalorhynchus hectori hectori*) and Māui dolphin (*C. h. maui*); 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	West Coast North Island; all coastal areas of South Island.
Focal localities	Areas where significant fisheries interactions are known to have occurred include waters over or close to the continental shelf surrounding the South Island and the west coast of the North Island.
Key issues	Improved estimation of Hector's and Māui dolphin spatio-temporal distributions affecting spatial overlap with fisheries, including in low-density areas; improved population size estimates for the South Coast South Island (SCISI) Hector's dolphin population; improved estimation of demographic parameters affecting potential population growth rates.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional subpopulation basis, or at finer spatial and temporal scales; improved understanding of non-fishery threats including disease.
MPI research (current)	PRO2017-12 *Hector's and Māui dolphin risk assessment to support review of the TMP;* PRO2016-09 *Abundance and distribution of Hector's and Māui dolphins on South Coast South Island;* PRO2015-04 *Addressing key information gaps for Māui dolphins*; PRO2014-01 *Improved estimation of the distribution of seabirds and marine mammals (cetaceans);*; PRO2013-09 *Population viability of Māui dolphins;* PRO2009-01C *Abundance, distribution and productivity of Hector's (and Māui) dolphins* (ECSI survey).
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): MIT2012-03 *Review of mitigation techniques in set net fisheries*; INT2013-01 *To understand the nature and extent of protected species interactions with New Zealand commercial fishing activities*; INT2013-03 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*; INT2013-04 *To review the data collected by fisheries observers in relation to understanding the interaction with protected species, and refine efficient protocols for future data collection*; Additional conservancy-level work including aerial and boat surveys in Taranaki, genetic sampling and necropsies of recovered animals.
Other research[1]	Otago University: Long term study of Hector's dolphins at Banks Peninsula, including distribution and abundance, survival rates, reproductive rates, movements, feeding ecology. Auckland University: Population monitoring of Māui dolphins and population genetics of Hector's and Māui dolphins. Massey University: Necropsy of recovered Hector's/Māui dolphins.
Related chapters/issues	Chapter 3 (SEFRA); Chapters 4–5 (sea lions and fur seals); Chapter 7 (common dolphins)

Note: This chapter has been updated for the 2017 AEBAR with recent population estimates and capture estimation and risk assessment results.

[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).

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 at risk of fishing-related mortality (e.g., Read et al. 2006, Reeves et al. 2013a, 2013b, Geijer & Read 2013).

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).

The latest DOC Marine Mammal Action Plan[3] (DOC MMPA; Suisted & Neale 2004) stated that actions required include:

- *'Prepare species plans for both Hector's and Maui's dolphins'*
- *'Consider preparation of Population Management Plans (PMP) for Hector's and Maui's dolphins in accordance with the legal process and the species plans.'*

However, to date no population management plan (PMP) has been produced for Hector's or Māui dolphin and no maximum allowable level of fishing-related mortality has been set. A draft threat management plan (TMP) for Hector's and Māui dolphin was 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 Hector's and Māui dolphin populations and outlining strategies to mitigate those threats. The stated goals of the TMP (DOC & MFish 2007) are:

- *'To ensure the long-term viability of Hector's and Maui's dolphins is not threatened by human activities; and*
- *'To further reduce impacts of human activities as far as possible, taking into account advances in technology and knowledge, and financial, social and cultural implications.'*

These goals were re-stated in the consultation paper on the Review of the Māui dolphin portion of the TMP published in 2012 (MPI & DOC 2012). The review of the Māui portion of the TMP provided a comprehensive overview of information relating to the biology, distribution, threats to, and management of Māui dolphins. To inform the review of the Māui dolphin TMP, a spatially explicit, semi-quantitative risk assessment was 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). The process involved expert panelists mapping dolphin distribution, identifying and characterising threats, scoring the likely impact of each threat, and subsequent quantitative analysis to estimate risk posed by threats. Where these remain relevant, the

[2] In this document, 'Hector's dolphin(s)' refers to the South Island subspecies (*Cephalorhynchus hectori hectori*), while '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.

[3] DOC has confirmed that the Marine Mammal Action Plan for 2005–10 still reflects DOC's priorities for marine mammal conservation.

results of this process are described in the relevant sections below.

A review of the full TMP will begin in 2018. The review will be informed by a 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 (MPI project PRO2017-12). The risk assessment will consider updated estimates of population size, updated demographic parameters affecting population growth and recovery potential, and improved estimates of the distribution of the dolphins to better estimate spatial overlap with threats, adapting methods described in Chapter 3. This information will be used to reassess the risk of fishing- and non-fishing-related threats across Hector's and Māui dolphin subpopulations, to evaluate the effectiveness of the management measures and monitoring programmes currently in place, and to consider the effects of alternative risk management scenarios where these may be required.

6.2 BIOLOGY

6.2.1 TAXONOMY

Hector's and Māui dolphin is one of four species in the genus *Cephalorhynchus*, which are all restricted to cool, temperate, coastal waters in the southern hemisphere. On the basis of morphological differences, and genetic information indicative of reproductive isolation, Hector's and Māui dolphin was divided into two subspecies; Hector's dolphin around the South Island (41˚S to 47˚S) and Māui dolphin, on the west coast of the North Island (WCNI, 36°S to 40°S; Baker et al. 2002). The reproductive isolation of the Māui subspecies is supported by a more recent genetic analysis with a larger sample size (Hamner et al. 2012a) despite genetic analyses having located four Hector's dolphins off the WCNI (Hamner et al. 2014).

6.2.2 DISTRIBUTION

Hector's dolphins are most frequently sighted on the west coast of the South Island (WCSI) between Jackson Bay and Kahurangi Point (Bräger & Schneider 1998, Rayment et al. 2011a), on the east coast (ECSI) between the Marlborough Sounds and Otago Peninsula (Dawson et al. 2004, MacKenzie & Clement 2014) and on the south coast (SCSI) between Toetoes Bay and Porpoise Bay and in Te Waewae Bay (Bejder & Dawson 2001, Dawson et al. 2004). Current population densities are lower in the intervening stretches of coast, e.g., Fiordland (Bräger & Schneider 1998), Golden Bay (Slooten et al. 2001) and the south Otago coast (Jim Fyfe, pers. comm.), resulting in a fragmented distribution. There is significant genetic differentiation among the west, east and south coast populations, with little or no gene flow connecting them (Pichler et al. 1998, Pichler 2002, Hamner et al. 2012a). The observed levels of genetic divergence over such small distances are unusual among cetaceans, especially considering the absence of geographical barriers (Pichler et al. 1998). These genetic differences are thought to result from individuals having small home ranges and high philopatry (Pichler et al. 1998, Bräger et al. 2002, Rayment et al. 2009b). For example, the mean lifetime alongshore home range of the 20 most frequently sighted dolphins at Banks Peninsula was 49.7 km (s.e.: 5.29; ranging from 13.60 km to 101.43 km for individual dolphins) for the period 1985 to 2006 (Rayment et al. 2009b).

Satellite tagging of three Hector's dolphins off the Banks Peninsula in 2004 indicated maximum distances between locations of 50.9 to 66.5 km over deployments lasting from four to seven months (Stone et al. 2005). For photo-identified dolphins, Rayment et al. (2009a) reported distances between extreme sightings for 53 dolphins ranging from 9.34 km to 107.38 km for the period 1985–2006.

Genetic testing of dolphins off the WCNI 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 raise the possibility of at least occasional long distance dispersal by Hector's dolphins (Hamner et al. 2012b). Although some of these dolphins were found in association with Māui dolphins there is currently no evidence of interbreeding (Hamner et al. 2014). Some of the Hector's dolphins sampled on the WCNI could not be unambiguously assigned to one of the three South Island Hector's dolphin populations leading Hamner et al. (2014) to raise the possibility that they may represent a hitherto unsampled population of Hector's dolphins, or indicate interbreeding between the ESCI and WCSI populations.

Māui dolphins are most frequently sighted between Maunganui Bluff and New Plymouth (Slooten et al. 2005, Du Fresne 2010, Hamner et al. 2012a, 2012b). Research surveys since 2003 have sighted Māui dolphins 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 on the WCNI from Dargaville to Wellington (DOC 2017a, 2017b, Hamner, pers. comm.); however there are doubts as to the provenance of a record of a Māui dolphin attributed to the Bay of Islands (Hamner, pers. comm.).

There are reported public sightings of Hector’s and Māui dolphins from all around the North Island coast, including the Bay of Islands, Hauraki Gulf, Coromandel Peninsula, Hawkes Bay, Wairarapa and Kapiti Coast (Baker 1978, Cawthorn 1988, Russell 1999, DOC 2017a). 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. It has also been suggested that Māui dolphin’s range has contracted off the WCNI in recent history coincident with a decline in abundance (MPI & DOC 2012).

Small-scale movements by Māui dolphins over up to 80 km of coastline have been revealed by repeated genetic sampling of the same individuals (mean distance between the two most extreme locations for the six individuals sampled at least three times = 35.5 km; s.e.: 4.03 km; Oremus et al. 2012).

Hector’s and Māui dolphin densities are highest close to the coast throughout the year. Bräger et al. (2003) used resource selection models to show that Hector’s dolphins have a preference for shallow, turbid waters. During systematic aerial surveys on the WCSI (Rayment et al. 2011a, MacKenzie & Clement 2016), east coast (MacKenzie & Clement 2014, Figure 6.2 and Figure 6.3), at Banks Peninsula (Rayment et al. 2010), in Cloudy and Clifford Bays (DuFresne & Mattlin 2009) and on the North Island west coast (Slooten et al. 2005) most sightings were in water depths less than 100 m (e.g., Figure 6.2 and Figure 6.3). Occasional sightings are made beyond the 100 m isobath (e.g., DuFresne & Mattlin 2009, MacKenzie & Clement 2014). Varying bathymetry among these locations meant that most sightings were within 6 nm offshore of the WCSI (Rayment et al. 2011a, MacKenzie & Clement 2016), yet extended at least out to 20 nm from the coast at Banks Peninsula (MacKenzie & Clement 2014). In both these areas, distance offshore best explained dolphin distribution, possibly due to declining prey availability with increasing distance from the coast (Rayment et al. 2010, 2011a). At Banks Peninsula, there was a significant seasonal difference in distribution, with a greater proportion of dolphins close to shore in summer than winter (Rayment et al. 2010, MacKenzie & Clement 2014), a conclusion consistent with nearshore boat-based surveys (e.g., Dawson & Slooten 1988, Bräger 1998) and passive acoustic monitoring (Rayment et al. 2009a). However, the furthest offshore sighting distances were similar in summer and winter (Rayment et al. 2010, MacKenzie & Clement 2014, MacKenzie & Clement 2016). From analysis of passive acoustic data, Dawson et al. (2013a) suggested that Hector’s dolphins’ use of an inner harbour site in Akaroa Harbour was greater than expected in winter, and that habitat selection was affected by time of day and state of the tide. No such seasonal difference in Hector’s dolphin distribution was detected during aerial surveys on the South Island west coast (Rayment et al. 2011a, MacKenzie & Clement 2016).

New aerial surveys of the South Coast South Island population are contracted for the summer of 2017-18 (MPI project PRO2016-09) to reduce uncertainty in the estimation of population size for the SCSI Hector’s dolphin subpopulation.

The highest density of Māui dolphins occurs inshore (within 4 nm of the coast) between Manukau Harbour and Port Waikato (Slooten et al. 2005, MPI & DOC 2012, Oremus et al. 2012). Sightings are occasionally made beyond 4 nm from the coast, extending at least to 7 nm offshore (Du Fresne 2010, Thompson & Richard 2012). Sightings of Māui dolphins have been made in three North Island harbours (Kaipara, Manukau and Raglan; see review in Slooten et al. 2005). Passive acoustic monitoring of these three harbours, in addition to Kawhia Harbour, revealed a low-level of episodic use of Kaipara and Manukau Harbours (Rayment et al. 2011b).

A map of Māui dolphin distribution[4] was developed as part of the Māui dolphin risk assessment (Currey et al. 2012). The distribution was generated via generalised additive modelling (Thompson & Richard 2012) of systematic survey data (Ferreira & Roberts 2003, Slooten et al. 2005, 2006, Scali 2006, Rayment & du Fresne 2007, Childerhouse et al. 2008, Stanley 2009, Hamner et al. 2012a) and modification to incorporate expert panel feedback regarding the alongshore, offshore and inshore extent (Figure 6.1; see Currey et al. 2012 for further details).

Preliminary results of work currently in progress under project PRO2014-01 suggest that Hector's and Māui dolphin distributions may be more effectively modeled, including potential seasonal changes, using multivariate models fitted to spatial environmental data layers, in particular satellite-derived estimates of turbidity, and/or modeled species distributions of preferred prey. This work will be expanded under project PRO2017-12 and used to inform the update of the Hector's and Māui dolphin Threat Management Plan in 2018.

6.2.3 FORAGING ECOLOGY

Miller et al. (2013) investigated the diet of Hector's and Māui dolphins through the examination of diagnostic prey remains in the stomachs of 63 incidentally captured and beach-cast animals. 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 mid-water and demersal species, particularly red cod (*Pseudophycis bachus*), ahuru (*Auchenoceros punctatus*), arrow squid (*Notodarus* sp.), sprat (*Sprattus* sp.), sole (*Peltorhamphus* sp.) and stargazer (*Crapatulus* sp.). 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 the juveniles of some species were targeted (Miller et al. 2013). The diets of Hector's dolphins from the South Island west and east coasts were significantly different, due largely to the importance of javelinfish (*Lepidorhynchus denticulatus*) on the west coast, and a greater consumption of demersal prey species on the east coast (Miller et al. 2013). Only two samples were derived from Hector's/Māui dolphins off the WCNI, containing only red cod, ahuru, sole and flounder (*Rhomboselea* sp.; Miller et al. 2013). The stomachs of the six smallest dolphins in the sample (standard length under 90 cm) contained only milk, while the next largest (99 cm standard length) contained milk and remains of arrow squid (Miller et al. 2013). Milk was not found in the stomachs of any dolphins longer than 107 cm (Miller et al. 2013).

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). Hector's dolphins are 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). The seasonal changes in distribution of Hector's dolphins at Banks Peninsula described above are presumed to be in response to seasonal movements of their prey species (Rayment et al. 2010), many of which migrate into shallower nearshore waters in the summer months (Paul 2000).

6.2.4 REPRODUCTIVE BIOLOGY

Incidentally captured and stranded Hector's dolphins have provided information on the life history and reproductive parameters of the species. Males reach sexual maturity between six and nine years of age, and females have their first calf between seven and nine years old (Slooten 1991). Examination of the ultrastructure of the teeth from these necropsied animals revealed that females live to at least 19 years (n = 33) and males (n = 27) to at least 20 (Slooten 1991). Photo-ID studies have provided additional data and revealed that the calving interval is two to four years (Slooten 1990) and that longevity is at least 22 years

[4] The map of Māui dolphin distribution was produced using data that included sightings of unknown subspecies identity (e.g., from aerial surveys). Hector's dolphins have been detected off the WCNI. However, they comprised just 4 of the 91 animals genetically identified within the area of mapped distribution since 2001 (two living females, one dead female, one dead male; Hamner et al. 2012a, 2013). The two living Hector's dolphins were found in association with Māui dolphins and three of four dolphins were found in or near Manukau Harbour, close to the core of Māui dolphin distribution (Figure 6.1). Given that the proportion of Hector's dolphins is likely to be small and there was no evidence to suggest that their inclusion would bias the distribution, the risk assessment proceeded with this map on the basis that it provided the best estimate of Māui dolphin distribution available.

(Rayment et al. 2009b, Webster et al. 2009). Gormley (2009) extended these analyses, estimating mean female fecundity of Hector's dolphins off Banks Peninsula at 0.205 female offspring per capita per annum (s.d.: 0.050) and mean age at first reproduction at 7.5 years (s.d.: 0.42).

Calves are typically born during spring and early summer, with neonatal length estimated to be 60–75 cm (Slooten & Dawson 1994). Calves stay with their mothers for at least one year, more usually two, and the mother does not appear to conceive again until the calf is independent (Slooten & Dawson 1994). Application of the growth models produced by Webster et al. (2010) to the diet data obtained by Miller et al. (2013) suggests that weaning occurs between one and two years of age. Growth is rapid and asymptotic length is reached in 5–6 years (Webster et al. 2010). Sexually mature adults usually fall within the range of 119–145 cm total length and at maturity females are approximately 10 cm longer than males (Slooten & Dawson 1994, Webster et al. 2010). In a sample of 66 female and 100 male known-age Hector's dolphins, the maximum total length measurements were 145 cm and 132 cm respectively (Webster et al. 2010). Māui dolphins are significantly longer than Hector's dolphins, with a maximum recorded total length of 162 cm (Russell 1999). 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. Using molecular sexing techniques, Oremus et al. (2012) found no evidence of sexual segregation in groups of fewer than eight Māui dolphins. The social organisation of Hector's dolphin groups is characterised by fluid association patterns, with little stability over periods longer than a few days (Slooten et al. 1993). Together with observations of sexual behaviour (Slooten 1990) and the relatively large testis size of males (Slooten 1991), this suggests that Hector's dolphins have a promiscuous mating system, in which males seek encounters with multiple females rather than attempting to monopolise them (Slooten et al. 1993).

These life-history characteristics mean that Hector's dolphins, like many other small cetaceans (Perrin & Reilly 1984), have a low intrinsic population growth rate. Using matrix population models, asymptotic population growth rate for Hector's dolphins was estimated to be -4.2 to +4.9% per year for survivorship schedules based on other mammals (Slooten & Lad 1991). The authors considered that a growth rate of 1.8% was a plausible 'best case' scenario for Hector's dolphin (Slooten & Lad 1991). Estimates of the intrinsic rate of increase from matrix models are sensitive to the particular parameters chosen (Slooten & Lad 1991, Gormley et al. 2012, Baker et al. 2013).

Updated information to estimate relevant demographic parameters for Hector's and Māui dolphins will be considered in the review of the TMP in 2018.

6.2.5 POPULATION BIOLOGY

The earliest survey-based abundance estimate for Hector's and Māui dolphin (3408 animals with a suggested range of 3000 to 4000) was obtained via small boat-based strip transects surveys (Dawson & Slooten 1988; Table 6.1). These surveys were primarily focused on assessing alongshore distribution rather than abundance. Consequently survey effort was concentrated within 800 m of shore and calibrated with a limited number of 5 nm offshore transects. Nationwide line transect surveys of Hector's and Māui dolphin were carried out between 1997 and 2004 (Dawson et al. 2004, Slooten et al. 2004, 2006). These resulted in a population estimate for Hector's dolphin around the South Island and offshore to 4 nm of 7270 (CV = 16%; Slooten et al. 2004) and for Māui dolphin of 111 (CV = 44%; Slooten et al. 2006).

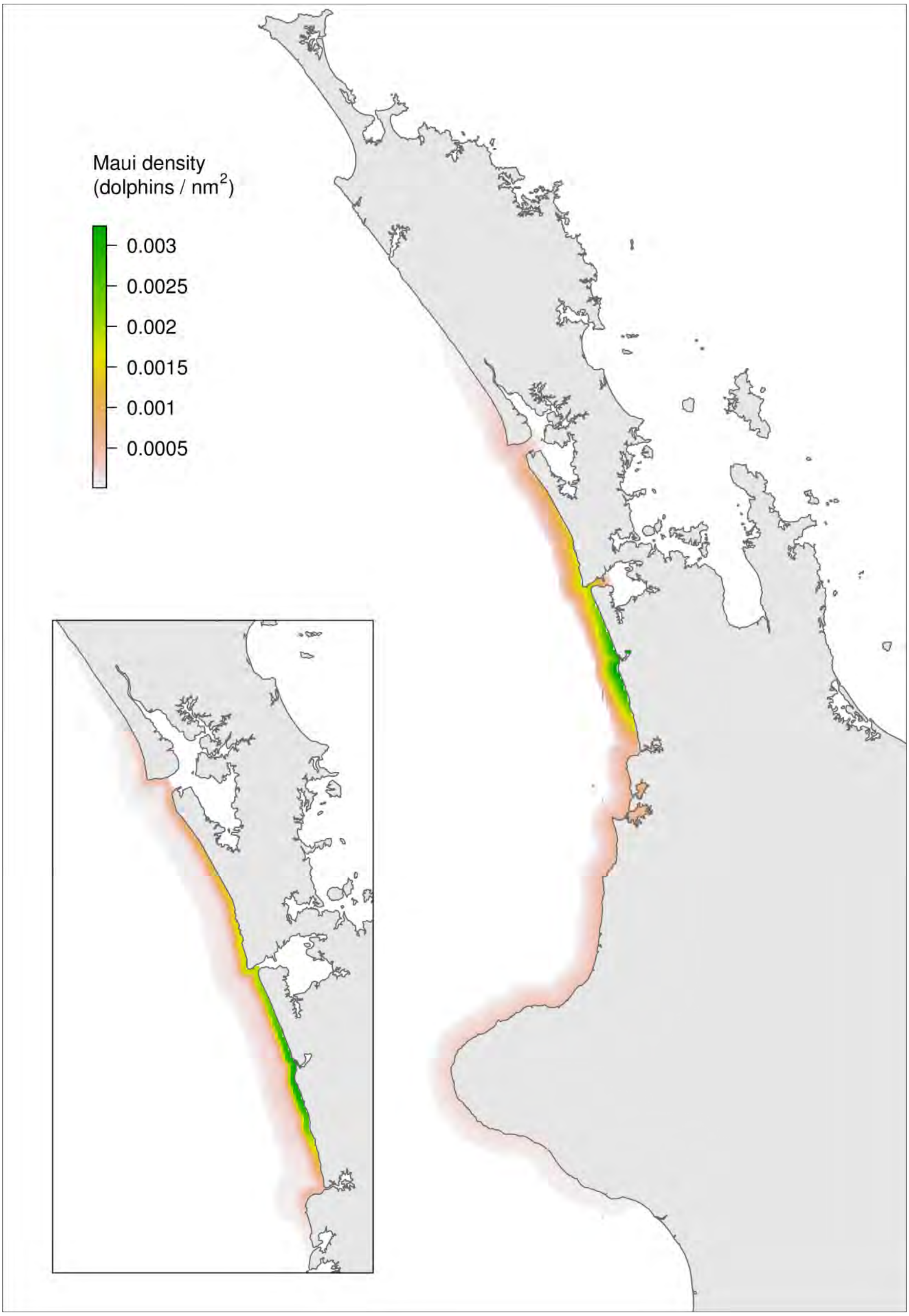

Figure 6.1: Māui dolphin distribution modelled from systematic survey data collected between 2000 and 2012 and modified to incorporate expert panel feedback (Currey et al. 2012). The inset depicts the modelled distribution prior to modification (Thompson & Richard 2012).

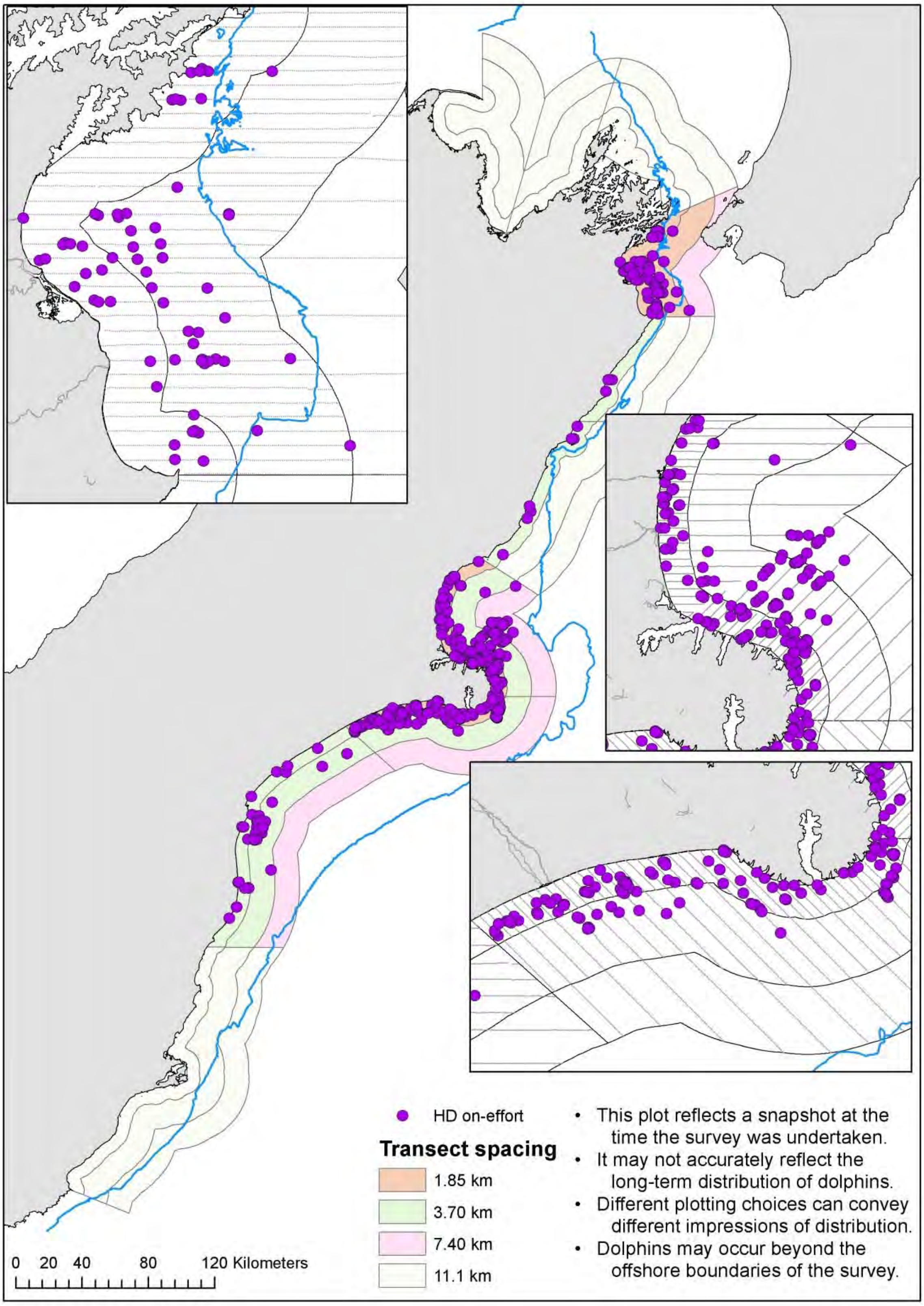

Figure 6.2: The distribution of all on-effort sightings of Hector's dolphins during the summer survey of the ECSI between 28 January and 13 March 2013. Reproduced from MacKenzie & Clement (2014).

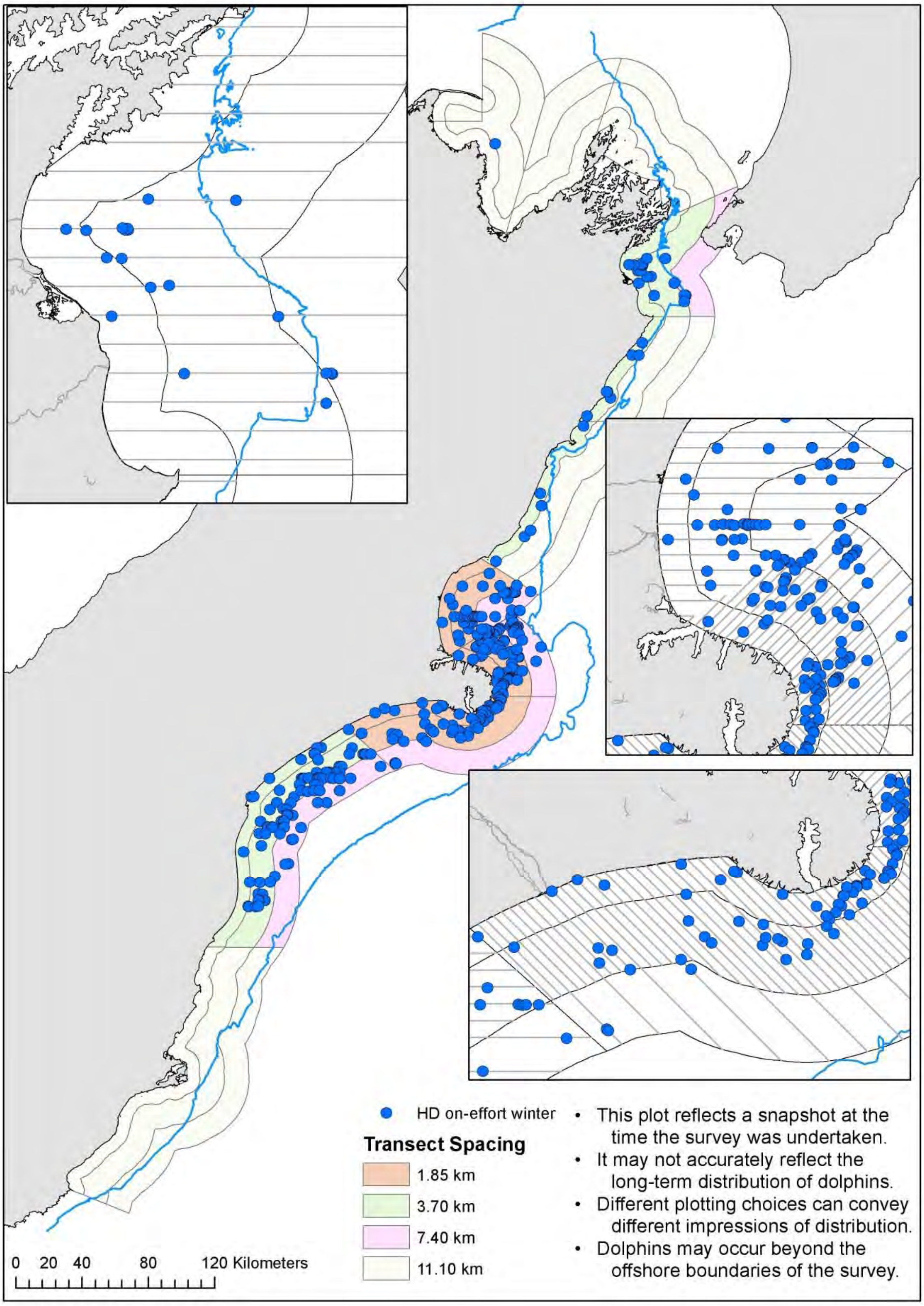

Figure 6.3: The distribution of all on-effort sightings of Hector's dolphins during the winter survey of the ECSI between 1 July and 18 August 2013. Reproduced from MacKenzie & Clement (2014).

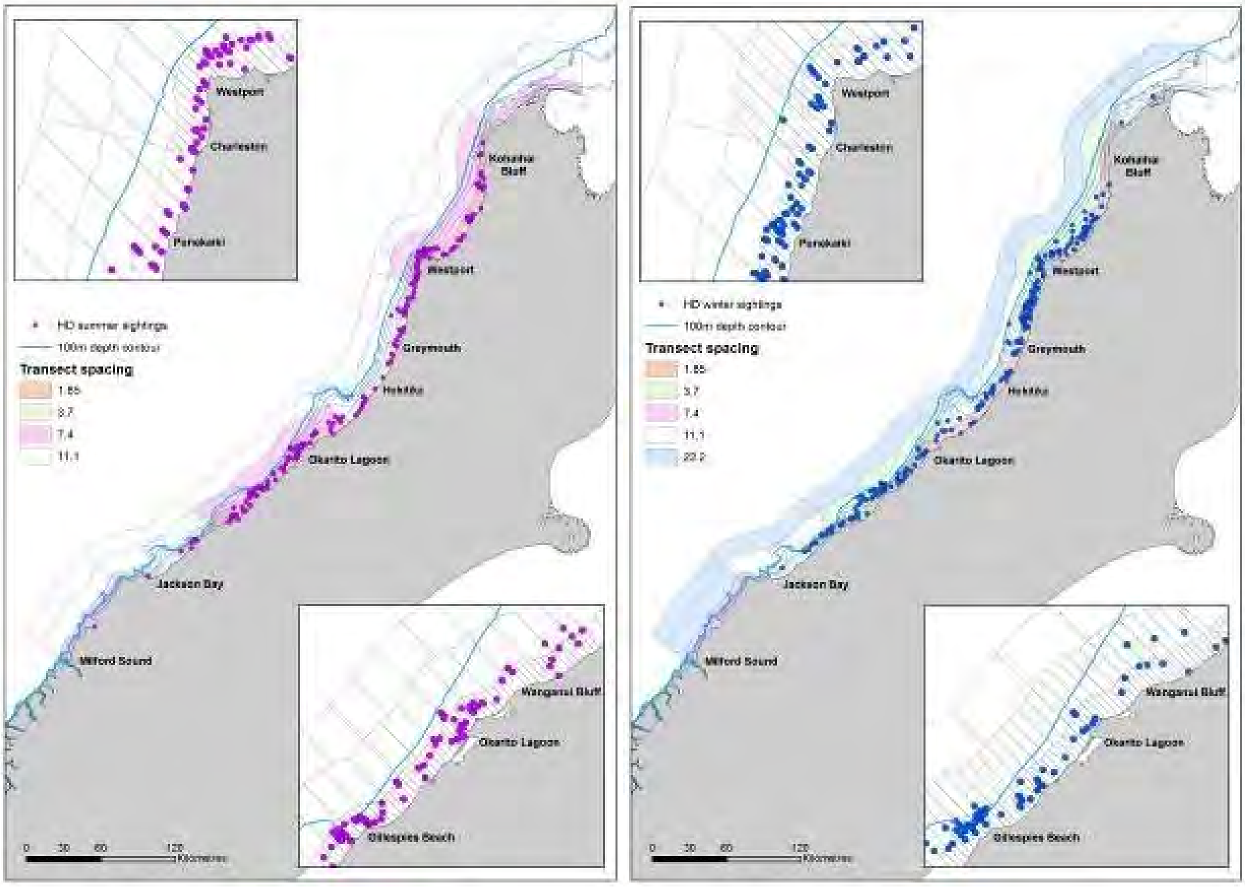

Figure 6.4: Hector’s dolphin distributions assessed from aerial line-transect surveys. Panels represent patterns for all on-effort Hector’s dolphin sightings in summer (left), and winter (right). Reproduced from MacKenzie & Clement (2016).

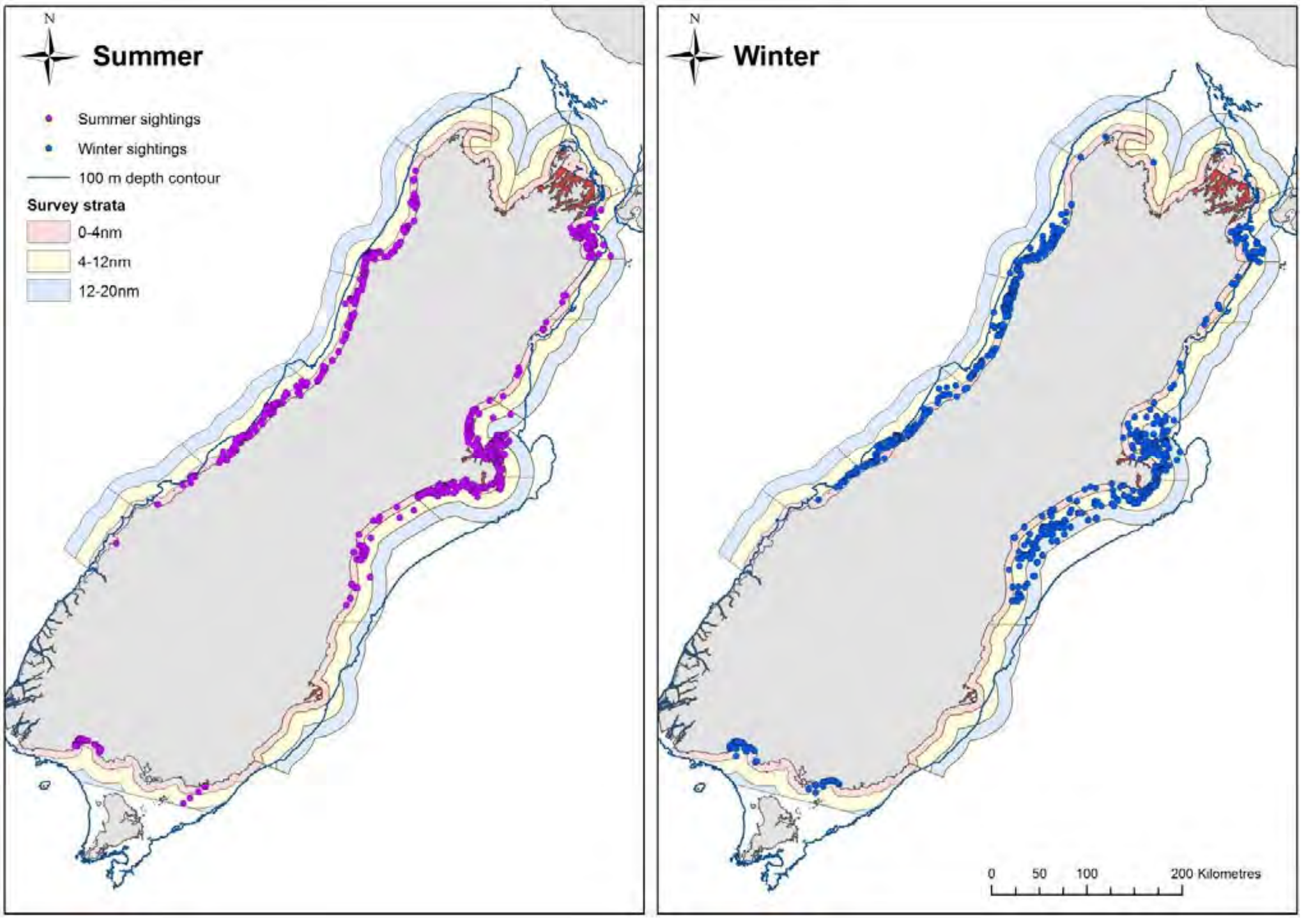

Figure 6.5: Hector's dolphin summer (left) and winter (right) sightings from the three separate abundance surveys: east coast (WCSI) completed 2015, east and north coast (ECSI) completed in 2013 and south coast (SCSI) completed in 2010. SCSI survey will be repeated in early 2018. Reproduced from MacKenzie & Clement (2016).

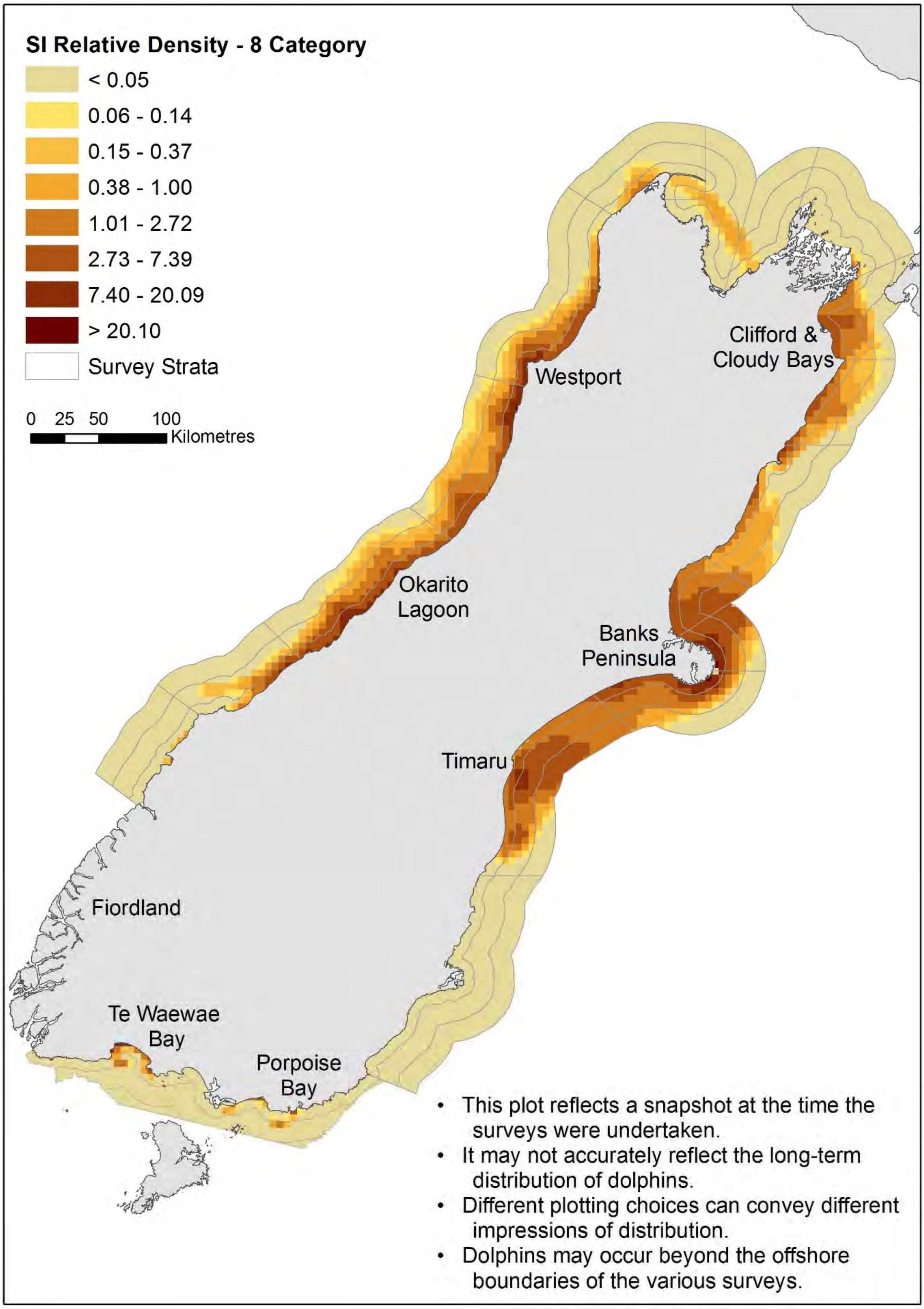

Figure 6.6: The South Island distribution of Hector's dolphin assessed from both summer and winter aerial line-transect surveys. Shows the pattern of the relative density of Hector's dolphins within 5 × 5 km grid cells generated from the Density Surface Models with eight categories. Reproduced from MacKenzie & Clement (2016).

Further aerial surveys focused on assessing seasonal and annual variation in distribution around Banks Peninsula (Rayment et al. 2010) and in distribution and abundance in Cloudy and Clifford Bays (DuFresne & Mattlin 2009). There have also been a number of photo-ID mark-recapture estimates focused on subpopulations of Hector's dolphin (Bejder & Dawson 2001, Gormley et al. 2005, Turek et al. 2013; Table 6.1) and genotype mark-recapture estimates of abundance for Māui dolphins and Hector's dolphins in Cloudy Bay (Hamner et al. 2012b, 2013, Baker et al. 2013, 2016b; Table 6.1). The genetic mark-recapture data yielded estimates of average annual population change for Māui dolphin of -0.13 (i.e., a 13% decrease p.a.; 95% c.i.: -0.40–+0.14) for the period 2001–07 (Baker et al. 2013), and -0.03 (95% c.i.: -0.11–+0.06) for the period 2001–11 (Hamner et al. 2012b). Baker et al. (2016b) estimated an abundance of N = 63 with 95% log-normal CL = 57, 75 for the population of Māui dolphins one year old and older. This estimate is comparable to, but slightly larger than the previous estimate of N = 55 (95% CL = 48, 69) based on the genotype surveys in 2010–11 (Hamner et al. 2012b).

Population trends have also been inferred for Māui dolphins via other methods, including linear regression of the natural logarithm of abundance estimates obtained using a variety of survey methods over the period 1985 to 2011 (-0.032; 90% c.i.: -0.057 to -0.006 for aerial and boat surveys; -0.037; 90% c.i.: -0.042 to -0.032 for boat surveys alone; Wade et al. 2012). Analysis of the Māui dolphin risk assessment expert panel's mortality scores yielded an estimated rate of population decline of 7.6% per annum (95% c.i.: = 13.8% decline to 0.1% increase; Currey et al. 2012). Across methods, estimates of Māui dolphin population trends indicate a high probability that the population is declining, with mean or median estimates suggesting a rate of decline at or above 3% per annum (Currey et al. 2012, Hamner et al. 2012b, Wade et al. 2012, Baker et al. 2013). Based on the more recent genetic mark-recapture data (Baker et al. 2016b), the best-fitting Pradel Survival and Lambda model estimated the annual rate of change to be 0.983 (95% c.i.: 0.940–1.028). Therefore, the Māui dolphin estimates suggest that the population declined by approximately 1.5–2% per year between 2001 and 2016; however, the decline was not confirmed with 95% confidence, as the upper confidence limits span a range up to a population increase of 3% per year (Baker et al. 2016b).

Recently, MPI-funded survey programmes (PRO2009-01A, PRO2009-01B, PRO2009-01C, PRO2013-06) were conducted to assess abundance and distribution of the SCSI, ECSI and WCSI populations of Hector's dolphin (Clement et al. 2011, MacKenzie et al. 2012, MacKenzie & Clement 2014, 2016). The SCSI 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 (PRO2009-01A, Clement et al. 2011). Seven dolphin groups were sighted during summer/autumn surveys and ten groups were observed in winter. Sightings data pooled across seasons were analysed using mark-recapture distance sampling (MRDS) with helicopter-based dive cycle observations used to correct for availability bias. SCSI Hector's dolphin abundance was estimated to be 628 dolphins (CV = 38.9%; 95% c.i.: 301–1311; Clement et al. 2011). MacKenzie & Clement (2016) reanalysed the SCSI survey data from 2014 and produced an annual average estimate for the SCSI of 238 (s.e.: 94; 95% c.i.: 113–503) based on revised figures for availability. Under project PRO2016-09, the SCSI survey will be repeated in early 2018 with higher sampling intensity in the nearshore strata to achieve a lower CV in the estimation of population size.

The ECSI program involved an initial design phase (PRO2009-01B, 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; PRO2009-01C; 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 (Figure 6.2 and Figure 6.3). Sightings data were analysed using MRDS and density surface modelling techniques to yield estimates of density and total abundance. The estimates of ECSI Hector's dolphin abundance were 9130 dolphins (CV = 19%; 95% c.i.: 6342–13 144) in summer 2012–13 and 7456 dolphins (CV = 18%: 95% c.i.: 5224–10 641) in winter 2013 (MacKenzie & Clement 2014). These estimates were obtained via model averaging four sets of MRDS results for each season; from two different datasets using different truncation distances and two methods of estimating availability (helicopter-based dive cycle and survey aircraft circle-backs). These estimates do not include harbours and bays that were outside of the survey region. MacKenzie & Clement (2016) reanalysed the ECSI survey data from 2014 and produced an annual average estimate for the ECSI of 8968 (s.e.: 1377; 95% c.i.: 6649–12 096), based on revised figures for availability.

A survey programme was specifically designed for sampling the WCSI population using two separate aerial surveys over summer 2014–15 and winter 2015 (MPI project PRO3013-06). The WCSI survey area (about 26 333 km^2 between Farewell Spit and Milford Sound) was stratified into six coastal sections, which were further divided into offshore substrata of 0–4 nm (inner), 4–12 nm (middle) and 12–20 nm (outer). This design was expected to encompass the offshore limits of Hector's dolphin distribution along the WCSI (MacKenzie & Clement 2016).

The WCSI Hector's dolphin summer abundance was estimated to be 5490 (CV: 26%; 95% c.i.: 3319–9079) 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).

Summer sightings results consisted of 250 Hector's dolphin groups (115 of which were seen by two observers) sighted within 0.3 km either side of the plane along 4001 km of transect lines. In winter, 272 Hector's dolphin groups (115 of which were seen by two observers) were sighted within 0.3 km either side of the plane along 4307 km of transect lines. Hector's dolphins were observed as far offshore as 12 km (6.5 nm) and 17.7 km (9.5 nm) in summer and winter, and in waters as deep as 160 m and 200 m, respectively. However, the majority of animals in both seasons occurred close to shore (less than 3 nm) and within relatively shallow depths (less than 40 m) (Figure 6.4; MacKenzie & Clement 2016).

Following the reanalysis of the ECSI and SCSI survey data, MacKenzie & Clement (2016) estimate the total Hector's population around the South Island (excluding sounds and harbours) to be 14 849 (CV: 11%; 95% c.i.: 11 923–18 492). This estimate is approximately twice as large as 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 a substantial number of dolphins estimated to be in offshore areas (greater than 4 nm) along ECSI that had not been extensively surveyed previously (Figures 6.4 and 6.5). Densities are similar along ECSI and WCSI (Figure 6.6; MacKenzie & Clement 2016).

For several years questions have been brought forward in International Whaling Commission (IWC) Sub-committee on Small Cetaceans concerning the methods used to derive abundance estimates of Hector's dolphins by New Zealand. The subcommittee agreed at the 2015 IWC meeting to review the abundance estimates intersessionally (International Whaling Commission 2016a). A formal process was established intersessionally following IWC procedures for such review and this included the creation of an Intersessional Expert Group (IEG) and an Intersessional Correspondence Group (ICG). The IEG consisted of independent experts who were asked to review the abundance methodology and estimates produced by MacKenzie and 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 an indepth 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. Thus, they believed that it follows 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).

Hector's dolphin is one of very few dolphin species for which estimates of survival are available. For long-lived species, a long time series of data is required to robustly estimate survival. The long-term photo-ID study at Banks Peninsula has facilitated several survival rate estimates since its inception in 1984 (Slooten et al. 1992, Cameron et al. 1999, Du Fresne 2004, Gormley et al. 2012). The most recent analysis utilises the most data and is therefore arguably the most powerful. Survival rate was estimated as 0.863 (95% c.i.: 0.647–0.971) for the period 1986–88, prior to the designation of the Banks Peninsula Marine Mammal Sanctuary, and 0.917 (95% c.i.: 0.802–0.984) from 1989–2006 after the designation (Gormley et al. 2012). Given the reproductive parameters detailed above, these survival rate estimates equate to a mean estimated population growth rate of 0.939 (95% c.i.: 0.779–1.025) pre-sanctuary and 0.995 (95% c.i.: 0.927–1.048) post-sanctuary (Gormley et al. 2012). In the post-sanctuary scenario, most of the uncertainty in the population growth estimate is due to uncertainty in the estimate of fecundity (Gormley et al. 2012).

Annual survival of the Māui dolphin has been estimated from the genotype mark-recapture data (Hamner et al. 2012b, Baker et al. 2013, 2016b). The best-fitting Pradel Survival and Lambda model for the data series, 2001–16, estimated the annual survival for age 1+ dolphins to be 0.888 (95% c.i.: 0.842–0.922; Baker et al. 2016b).

Where mark-recapture data are available, population models of Māui and Hector's dolphin subpopulations will be used to inform the update of the Hector's and Māui dolphin TMP in 2018.

6.2.6 KNOWN AND POTENTIAL THREATS

Fishing-related mortality is known to be a potentially serious threat to Hector's and Māui dolphins (DOC & MFish 2007, MPI & DOC 2012). Fisheries risk is described in Section 6.4, below.

Non fishery threats have also been observed but are difficult to quantify. There has been one confirmed death due to boat strike since 1921, a Hector's dolphin calf in Akaroa harbour in 1999 (Stone & Yoshinaga 2000, DOC 2017a). Other known sources of mortality include predation by sharks (e.g., Cawthorn 1988), disease (e.g., Roe et al 2013) and separation of calves from their mothers (DOC 2017a), possibly exacerbated by extreme weather conditions (DOC & MFish 2007, MPI & DOC 2012).

The presence of tourist vessels has been demonstrated to cause behavioural changes (Bejder et al. 1999, Martinez et al. 2012), as has underwater noise. There are potential negative effects due to bioaccumulation of organochlorines and heavy metals (reviewed by Slooten & Dawson 1994). Stockin et al. (2010) reported elevated levels of PCBs and organochlorine pesticides in the tissues of Hector's and Māui dolphins, but noted that no PCB concentrations were over the threshold considered to have immunological and reproductive effects. Additionally, both subspecies face pressures placed on coastal habitat through activities such as aquaculture, seabed mining, dredging and tidal energy installations (DOC & MFish 2007, Currey et al. 2012, MPI & DOC 2012).

A comprehensive list of the threats posed to Māui dolphins was produced as part of the spatially explicit, semi-quantitative risk assessment (Currey et al. 2012). The expert panel was asked to identify, analyse and evaluate all potential threats to Māui dolphins. Working from a previously established list of 47 potential threats to Hector's dolphins from the Hector's and Māui dolphin TMP (DOC & MFish 2007), the expert panel assessed 23 threats potentially relevant to Māui dolphins (i.e., present within their established distribution) in terms of whether these were likely to affect population trends within the next five years (Table 6.2). For each of these threats, the expert panel provided estimates of the number of Māui dolphin mortalities per year (Table 6.3).

The panel process resulted in estimated numbers of Māui dolphin mortalities from commercial set net fisheries of 2.33 (95% c.i.: 0.02–4.26) per annum, with spatial disaggregation of the estimates indicating that Māui dolphins are exposed to the greatest level of risk from set net fisheries in the area of the northern Taranaki coastline out to 7 nm offshore, and at the entrance to the Manukau Harbour. Subsequent interim measures banned set-net fishing within 2 nm of the Taranaki coast (between Pariokariwa Point and Hawera) and required full observer coverage of commercial set net fishing out to 7 nm. No Māui dolphins have been captured or sighted by observers in the Taranaki set-net fishery since observer coverage began in July 2012.

The expert panel's assessment of mortalities can be treated as testable hypotheses reflecting the limitations of available knowledge at that time and should be updated using new information. In particular, Roe et al.'s (2013) finding that 2 of 3 Māui dolphins tested in the period 2007 to 2011 had died as a result of *Toxoplasma gondii* infection, possibly as a result of run-off from terrestrial sources, indicates that the panel results (Table 6.3) may have underestimated mortality from this source. Roe et al. (2013) note that toxoplasmosis may have other effects beyond direct mortality and could be an important cause of neonatal loss. New work to investigate the risk of toxoplasmosis to Hector's and Māui dolphins is ongoing (W. Roe, pers. comm.) and will inform the update of the TMP in 2018.

6.2.7 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 and projected decline of greater than 80% over three generations, and there being fewer than 250 mature individuals remaining (Reeves et al. 2013a). Critically Endangered is the most threatened status before 'Extinct in the Wild'. Hector's dolphin is classified by the IUCN as Endangered under criterion A4d[6] due to an ongoing and projected decline of greater than 50% over three generations (Reeves et al. 2013b).

Under the New Zealand Threat Classification System (Baker et al. 2016a), Māui dolphin is classified as Nationally Critical, the most threatened status, under criterion A(1), with the qualifier Conservation Dependent (CD) [7] and Hector's dolphin as Nationally Endangered, the second most threatened status, under criterion C(1/1), with the qualifier Conservation Dependent (CD).[8]

Table 6.1: Abundance estimates for Hector's and Māui dolphin. N = estimated population size. * applies to individuals more than 1 year of age and includes two individuals genetically identified as Hector's dolphins. [Continued on next pages]

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
1984–85	Hector's and Māui dolphin	North and South Islands	Small boat based strip-transect	Distance sampling	3 408		3 000–4 000	Dawson & Slooten 1988
1989–97	Hector's dolphin	Banks Peninsula	Photo-ID	Mark-recapture	1 119	0.21	744–1682	Gormley et al. 2005
1995–97	Hector's dolphin	Porpoise Bay	Photo-ID	Mark-recapture	48		44–55	Bejder & Dawson 2001

[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 (d, 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 Endangered' under criterion C(1/1)when evidence indicates that the total population size is 1000–5000 mature individuals and there is an ongoing or predicted decline of 50–70% in the total population due to existing threats, taken over the next 10 years or three generations, whichever is longer (Townsend et al. 2008).

Table 6.1 [Continued]:

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
1997–98	Hector's dolphin	Motunau–Timaru (0–4 nm)	Boat based line-transect	Distance sampling	1 198	0.27	848–1 693	Dawson et al. 2004
1998–99	Hector's dolphin	Timaru–Long Point (0–4 nm)	Boat based line-transect	Distance sampling	399	0.26	279–570	Dawson et al. 2004
1999–2000	Hector's dolphin	Farewell Spit–Motunau (0–4 nm)	Boat based line-transect	Distance sampling	285	0.39	137–590	Dawson et al. 2004
2000–01	Hector's dolphin	Farewell Spit–Milford Sound (0–4 nm)	Aerial line-transect	Distance sampling	5 388	0.21	3 613–8 034	Slooten et al. 2004
2001–07	Māui dolphin	Kaipara Harbour–Tirua Point	Biopsy	Mark-recapture	59		19–181	Baker et al. 2013
2004	Māui dolphin	Maunganui Bluff–Pariokariwa Point (0–4 nm)	Aerial line-transect	Distance sampling	111	0.44	48–252	Slooten et al. 2006
2004–05	Hector's dolphin	Te Waewae Bay	Photo-ID	Mark-recapture	251 (autumn)	0.16	183–343	Green et al. 2007
					403 (summer)	0.12	280–488	
2006–09	Hector's dolphin	Cloudy and Clifford Bays (100 m contour)	Aerial line-transect	Distance sampling	951 (summer)	0.26	573–1 577	DuFresne & Mattlin 2009
					927 (autumn)	0.30	520–1 651	
					315 (winter)	0.31	173–575	
					188 (spring)	0.33	100–355	
2010	Hector's dolphin	Puysegur Point–Nugget Point (100 m contour)	Aerial line-transect	Distance sampling	628	0.39	301–1 311	Clement et al. 2011
2010 (reanalysis)	Hector's dolphin	Puysegur Point–Nugget Point (100 m contour)	Aerial line-transect	Distance sampling	238		113-503	MacKenzie & Clement 2016
2010–11	Māui dolphin	Kaipara Harbour–New Plymouth	Biopsy	Mark-recapture	57*		49–71	Hamner et al. 2012b
2010–11	Hector's dolphin	Taiaroa Head–Cornish Head	Photo-ID	Mark-recapture	42	0.41	19–92	Turek et al. 2013
2011–12	Hector's dolphin	Cloudy Bay	Biopsy	Mark-recapture	272	0.12	236–323	Hamner et al. 2013

Table 6.1 [Continued]:

Sampling period	Subspecies	Survey area	Survey method	Analysis method	N	CV	95% c.i.	Reference
2012–13	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	9 130 (summer)	0.19	6 342–13 144	MacKenzie & Clement 2014
					7 456 (winter)	0.18	5 224–10 641	
2012–13 (reanalysis)	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	9 728 (summer)	0.17	7 001–13 517	MacKenzie & Clement 2016
					8 208 (winter)	0.27	4 888–13 785	
2014–15	Hector's dolphin	Farewell Spit–Nugget Point (0–20 nm)	Aerial line-transect	Mark-recapture distance sampling	5 490 (summer)	0.26	3 319–9 079	MacKenzie & Clement 2016
					5 802 (winter)	0.21	3 879–8 679	
2015–16	Māui dolphin	Kaipara Harbour–New Plymouth	Biopsy	Mark-recapture	63		57–75	Baker et al. 2016b

6.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Coastal cetaceans are impacted by incidental capture in fisheries throughout the world (Read et al. 2006, Read 2008, Reeves et al. 2003). Read et al. (2006) estimated that global incidental captures of cetaceans exceeded 270 000 p.a. in the mid-1990s and that more than 95% of incidental captures occurred in set nets. Hector's and Māui dolphins are endemic to New Zealand and hence discussion of fisheries interactions for the species is detailed below under state of knowledge in New Zealand.

6.4 STATE OF KNOWLEDGE IN NEW ZEALAND

It is widely accepted that incidental mortality in coastal fisheries, notably set nets and to a lesser extent trawls, is the most significant threat to Hector's and Māui dolphins (MFish & DOC 2007, Slooten & Dawson 2010, Currey et al. 2012; see (Table 6.3). Hector's and Māui dolphins have been caught in inshore commercial and recreational set net fisheries since at least the early 1970s (Taylor 1992). Incidental mortalities have been documented throughout the species' range (Table 6.4). Beach-cast carcasses are frequently reported by members of the public, with the greatest number of reports coming from the east coast of the South Island (DOC 2017a). The numbers reported in the DOC Incident Database are not representative of the total magnitude or relative scale of incidental capture (DOC 2017a, Slooten 2013) because 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. Carcass reporting is also likely to be correlated with proximity to major population centres and thoroughfares. The information in the incident database (Table 6.4) provides only a biased indication of incidental captures. It is clear from this information, however, that incidental captures 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).

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).

Nineteen individual Hector's dolphins were reported caught in trawl fisheries between 1921 and 2008 and one since 2008 (Table 6.4; DOC 2017a). The first report of incidental capture in the commercial trawl fishery dates back to 1973 (Baker 1978). Note however that in the application of fisheries risk assessment methods used by MPI, only fishing effort that has been independently observed is used to estimate capture rates per encounter with fishing.

There have been three known incidents of Hector's dolphins becoming entangled in buoy lines of pots set for crayfish (*Jasus edwardsii*), all from Kaikōura (DOC & MFish 2007, DOC 2017a). From July 2008 to November 2017, there have been seven additional incidents of known entanglements in commercial set nets (six from the ECSI, one from WCNI) and five incidents of probable entanglements (three from ECSI, one from WCSI and one from the north coast of the South Island). These additional data are valid as of December 2017 (Table 6.4; DOC 2017a).

There are discrepancies between the data presented in the DOC Incident Database (2017a) and elsewhere in the published literature. Dawson (1991) collated reports of known incidental captures in Canterbury between 1984 and 1988 based on interviews with fishers. The minimum estimate of incidental captures in commercial set nets was 200 and in amateur nets was 24 (Dawson 1991), both of which are appreciably higher than the numbers presented in Table 6.4. These interview estimates were reviewed by Voller (1992) who reported a total of 112 entanglements in commercial nets from Timaru to Motanau in the period 1984–88 and attributed the difference from Dawson's results to the assumptions made about information provided by three individuals. There are a number of reasons why the people who were interviewed multiple times may have provided different information regarding incidental captures.

Table 6.2: Characterisation of threats evaluated as relevant to Māui dolphins and likely to affect population trends within the next five years. Reproduced from Currey et al. (2012). [Continued on next page]

Threat class	Threat	Mechanism	Type	Population component(s) affected
Fishing	Commercial trawl	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Commercial set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Recreational set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Recreational driftnet	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Customary set net	Incidental capture, cryptic mortality	Direct	Juvenile or adult survival
	Trophic effects	Competition for prey, changes in abundance of prey and predator species	Indirect	Fecundity, juvenile or adult survival
	Vessel noise: displacement, sonar	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
Vessel traffic	Boat strike	Physical injury/mortality	Direct	Juvenile or adult survival
	Disturbance	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
Pollution	Agricultural run-off	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
	Industrial run-off	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
	Plastics	Compromising dolphin health, ingestion and entanglement	Both	Fecundity, juvenile or adult survival
	Oil spills	Compromising dolphin health, ingestion (direct and prey) and inhalation	Both	Fecundity, juvenile or adult survival
	Trophic effects	Changes in abundance of prey and predator species	Indirect	Fecundity, juvenile or adult survival
	Sewage and stormwater	Compromising dolphin health, habitat degradation, trophic effects	Indirect	Fecundity, juvenile or adult survival
Disease	Natural	Compromising dolphin health	Both	Fecundity, juvenile or adult survival
	Stress-induced	Compromising dolphin health	Both	Fecundity, juvenile or adult survival
	Domestic animal vectors	Compromising dolphin health	Both	Fecundity, juvenile or adult survival

Table 6.2 [Continued]:

Threat class	Threat	Mechanism	Type	Population component(s) affected
Small population effects	Stochastic and Allee effects	Increased susceptibility to other threats	Indirect	Fecundity, juvenile or adult survival
Mining and oil activities	Noise (non-trauma)	Displacement from habitat, masking biologically important behaviour	Indirect	Fecundity, juvenile or adult survival
	Noise (trauma)	Compromising dolphin health	Direct	Fecundity, juvenile or adult survival
	Pollution (discharge)	Compromising dolphin health	Indirect	Fecundity, juvenile or adult survival
	Habitat degradation	Displacement from habitat, reduced foraging efficiency, trophic effects	Indirect	Fecundity, juvenile or adult survival

Table 6.3: Estimated number of Māui dolphin mortalities per year, the risk ratio of estimated mortalities to PBR and the likelihood of exceeding PBR for each threat, as scored by the expert panel. Individual threat scores were bootstrap resampled from distributions specified by the panel members and aggregated to generate medians and 95% confidence intervals. Modified from Currey et al. (2012).

Threat	Estimated mortalities		Risk ratio		Likelihood of exceeding PBR
	Median	95% c.i.	Median	95% c.i.	Median percentage
Fishing*	4.97	0.28–8.04	71.5	3.7–143.6	100.0
Commercial set net fishing*	2.33	0.02–4.26	33.8	0.3–74.3	88.9
Commercial trawl fishing*	1.13	0.01–2.87	16.7	0.1–48.5	88.9
Recreational/customary set net fishing	0.88	0.02–3.14	12.8	0.3–50.9	88.7
Recreational driftnet fishing	0.05	0.01–0.71	0.7	0.1–10.9	41.3
Trophic effects of fishing	0.01	<0.01–0.08	0.1	<0.1–1.2	4.7
Vessel noise/disturbance from fishing	<0.01	<0.01–0.10	<0.1	<0.1–1.6	9.0
Mining and oil activities	0.10	0.01–0.46	1.5	0.1–7.4	61.3
Habitat degradation from mining and oil activities	0.03	<0.01–0.17	0.4	<0.1–2.7	26.4
Noise (non-trauma) from mining and oil activities	0.03	<0.01–0.23	0.5	<0.1–3.6	28.6
Noise (trauma) from mining and oil activities	0.01	<0.01–0.13	0.2	<0.1–2.0	8.8
Pollution (discharge) from mining and oil activities	<0.01	<0.01–0.13	0.1	<0.1–2.2	13.4
Vessel traffic	0.07	<0.01–0.19	1.0	0.1–3.1	47.8
Boat strike from all vessels	0.03	<0.01–0.10	0.5	<0.1–1.6	17.9
Vessel noise/disturbance from other vessels	0.02	<0.01–0.12	0.3	<0.1–1.9	14.4
Pollution	0.05	<0.01–0.36	0.8	<0.1–5.9	40.2
Oil spills	0.02	<0.01–0.15	0.4	<0.1–2.4	20.4
Agricultural run-off	<0.01	<0.01–0.12	<0.1	<0.1–1.9	9.6
Industrial run-off	<0.01	<0.01–0.11	<0.1	<0.1–1.7	7.6
Sewage and stormwater	<0.01	<0.01–0.11	<0.1	<0.1–1.6	7.3
Trophic effects of pollution	<0.01	<0.01–0.06	<0.1	<0.1–0.9	2.1
Plastics	<0.01	<0.01–0.01	<0.1	<0.1–0.1	<0.1
Disease	<0.01	<0.01–0.36	<0.1	<0.1–5.5	29.5
Stress-induced diseases	<0.01	<0.01–0.35	<0.1	<0.1–5.2	20.7
Domestic animal diseases	<0.01	<0.01–0.07	<0.1	<0.1–1.1	3.9
Total	5.27	0.97–8.39	75.5	12.4–150.7	100.0

*Note that since the completion of the Marine Mammal Risk Assessment (Abraham et al. 2017 and Section 6.4.4, below) subjective estimates of commercial fisheries risk from Currey et al. (2012) are no longer considered best available information, but are retained for completeness until a comprehensive multi-threat assessment is available. Information in this table will be replaced following the update of the TMP in 2018.

Table 6.4: Fishing-related cause of death of Hector’s and Māui dolphins 1921–2008 and 2008–16 by 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.

	Cause of death	ECSI	WCSI	SCSI	WCNI	Unknown population
From 1921 to 2008						
Known entanglement[9]	Commercial set net	41	2	0	0	2
	Recreational set net	12	9	0	0	0
	Unknown set net	15	6	0	2	1
	Trawl net	15	4	0	0	0
Probable entanglement[10]	Commercial set net	0	0	0	0	0
	Recreational set net	0	0	0	0	0
	Unknown set net	1	4	0	0	0
	Unknown net	8	4	1	1	0
Possible entanglement[11]	Commercial set net	0	0	0	0	0
	Recreational set net	1	0	0	0	0
	Unknown set net	16	10	0	0	0
	Unknown net	16	7	1	2	0
From July 2008 to December 2017						
Known entanglement[12]	Commercial set net	5	0	0	1	0
	Recreational set net	1	1	0	0	0
Probable entanglement[13]	Recreational set net	3	0	0	0	1
	Unknown set net	0	1	0	0	0
Possible entanglement[14]	Commercial set net	1	0	0	0	0

[9] Animal was known (from incident report) to have been entangled and died.
[10] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[11] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[12] Animal was known (from incident report) to have been entangled and died.
[13] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[14] As read from pathology report, or presence of net marks on body and a mention of this in incident report.

Table 6.5: Summary of observed inshore set-net and trawl events, and Hector's and Māui dolphin captures, 1997–2012 (see also Baird & Bradford 2000, Blezard 2002, Fairfax 2002, Rowe 2009, 2010, Ramm 2010, 2012a, 2012b). Observed fishing effort, measured in kilometres of net set, or number of trawl tows. Fishing effort numbers are taken from linked fisher reports where possible. The inshore trawl effort is defined as being vessels less than 28 m, targeting flat fish (FLA, LSO, ESO, SFL, YBF, FLO, GFL, TUR, BFL, PAD) or inshore species (TAR, SNA, GUR, RCO, TRE, JDO, STA, ELE, LEA, QSC, MOK, SCH, SPO, BCO, RSK, HPB, LDO). FMAs include areas within and outside Hector's and Māui dolphin distribution (within: 3, 5, 7, 8 and 9; outside: 1, 2 and 10).

Fishing year	Set net					Inshore trawl			
	Areas (FMAs)	Total effort (sets)	Total effort (kms)	Observed effort (%)	Observed captures	Areas (FMAs)	Effort (tows)	Observed effort (%)	Observed captures
1997–98	3	214	260	0.87	8	3, 5, 7, 10	403	0.5	1
1998–99						2	15	0.02	0
1999–00						2, 3, 9	24	0.04	0
2000–01	3	535	24	0.08	0	2, 3	47	0.08	0
2001–02						1, 3, 9	25	0.04	0
2002–03						1	1	0	0
2003–04						3	4	0.01	0
2004–05						3	2	0	0
2005–06	3, 5, 7, 8	458	139	0.57	0	2, 7, 9	49	0.08	0
2006–07	3, 5, 7, 8	413	167	0.69	1	1, 3, 5, 7, 8, 9	260	0.46	0
2007–08	3, 5, 7, 8, 9	821	295	1.4	1	1, 3, 7, 8, 9	102	0.22	0
2008–09	3, 5, 7, 9	1 829	504	2.41	1	1, 3, 5, 7, 8, 9	1 682	3.46	0
2009–10	1, 3, 5, 7	1 927	580	2.61	2	1, 3, 5, 7	788	1.47	0
2010–11	2, 3	514	174	0.81	0	1, 2, 5, 7, 8	744	1.52	0
2011–12	7, 8, 9	161	75	0.37	0	1, 3, 7	328	0.67	0

6.4.1 OBSERVED FISHERIES INTERACTIONS

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 caught in Area 022 with 18 dolphins (CV = 38%)[15] estimated caught in Areas 020 and 022 combined (Baird & Bradford 2000). The authors stress that the preceding estimates are of dolphins caught, and not necessarily of mortalities (Baird & Bradford 2000). Note also that these estimates are from Statistical Areas containing the Banks Peninsula Marine Mammal Sanctuary, 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).

The spatial distribution of inshore set-net and trawl fishery effort is presented in Figure 6.7. The level of observation of inshore set-net fisheries since 1998 has been low (Table 6.5). 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 (see below). While this analysis has not been reviewed by the MPI Aquatic Environment Working Group (AEWG), a similar analysis extrapolating a capture rate estimated around Kaikōura across the ECSI was previously presented to an

[15] 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.).

AEWG and rejected given the unrepresentative nature of the observer coverage.

In the 2012–13 year, the inshore set-net fishery operating in Statistical Areas 022 and 024 was observed by human observers and electronic monitoring. During that time, at least two Hector's dolphins were captured, with one released alive.

Hector's dolphin captures in trawl nets include an individual caught in a trawl targeting red cod (*Pseudophycis bacchus*) in area 022 in 1997–98 (Starr & Langley 2000) and the capture of three Hector's dolphins in a trawl in Cloudy Bay in 2006 (DOC & MFish 2007). 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. While there have been ongoing attempts to increase the level of observer coverage in inshore trawl fisheries, it remains low (Table 6.5). A simple extrapolation using capture rate and total fishing effort suggests that the number of dolphins caught in trawl fisheries could be as high as the number caught in set nets (Slooten & Davies 2012).

In addition to data gathered by human observers, electronic monitoring of inshore set net and trawl fisheries has been trialed (McElderry et al. 2007). The trial monitored 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 (McElderry et al. 2007), reflecting a similar catch rate to previous estimates. Observers and electronic monitoring were deployed in the Timaru set-net fishery in 2012–13 and observers were deployed again in 2013–14. One confirmed and one probable capture of Hector's dolphins were observed (Archipelago 2013, MPI 2016).

Until recently, no attempt to quantify total captures of Māui dolphins in set nets or trawls using population-specific observer data was possible, because historical observer coverage was low and at low population sizes even very rare events may constitute considerable risk. In the absence of an empirical estimate, the government relied instead on a semi-quantitative risk assessment relying on estimates provided by a panel of nine domestic and international experts (Currey et al. 2012). The panel estimates attributed 95.5% of the mortality risk to fishing-related activities and 4.5% to non-fishing related threats, with captures in commercial set nets assessed as posing the greatest risk (Table 6.3; Currey et al. 2012). This risk assessment was conducted before the introduction of interim measures off the west coast of the North Island in 2012, so reflects historical rather than current risk. Since the introduction of interim measures, commercial set-net vessels have been required to carry an MPI observer when operating off the Taranaki coastline from 2 to 7 nm offshore between Pariokariwa Point and Hawera (i.e., outside the existing set net closure area). There have been no observed captures and no observations of dolphins in this area since July 2012 when this observer coverage began.

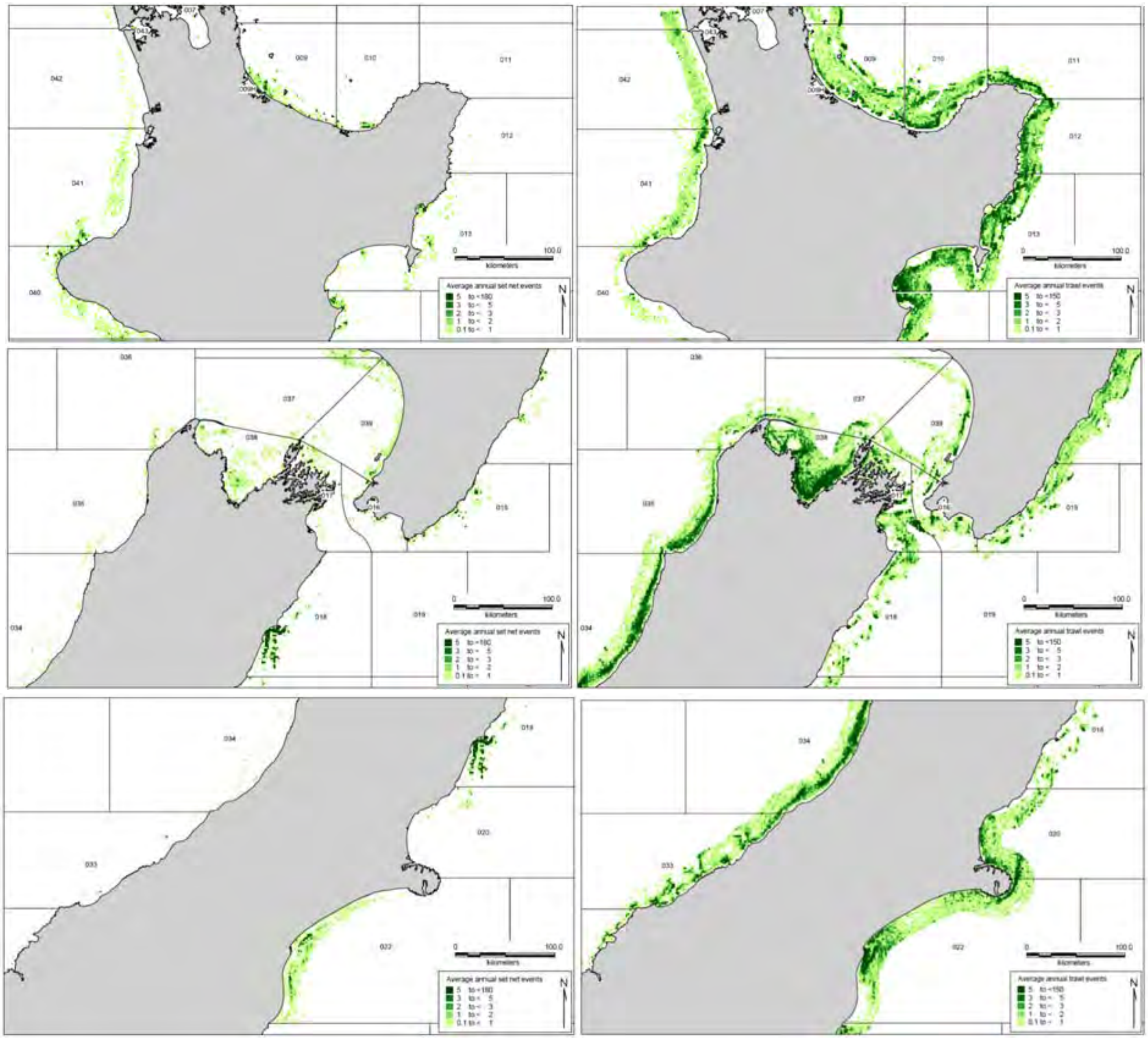

Figure 6.7: The distribution of set-net (left) and trawl (right) fishing events 2007–08 to 2009–10 to show the general spatial pattern of fishing activity. The annual average number of events (start positions) is shown for each 1 nm grid cell for events reporting coordinates (about 33% of set-net events, almost 100% of trawl events). Black lines show general statistical areas. Fishing returns are subject to occasional errors in method codes and coordinates; where possible, these errors have been corrected.

6.4.2 MANAGEMENT OF FISHERIES INTERACTIONS

Broadly, there are three potential solutions to managing incidental captures: gear modifications, mortality limits and spatial closures (Dawson & Slooten 2005). Gear modifications aimed at reducing cetacean captures 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 make its presence more obvious (Dawson et al. 2013b). Setting mortality limits involves determining a level of mortality that is sustainable (e.g., Wade 1998), and closing the fishery when it is reached. Both these approaches have been used as Hector's dolphin management tools. Most ECSI set-net fishermen voluntarily use pingers under a Code of Practice (Southeast Finfish Management Company 2000), and for a period had an annual mortality limit of three Hector's dolphins for the Canterbury gillnet fishery, which is no longer in effect (Hodgson 2002). Although 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), cetaceans can become habituated to the presence of pingers (Cox et al. 2001) and fishers do not necessarily deploy them correctly in real fisheries (Cox et al. 2007, Dawson et al. 2013b). Further, a trial reporting that 10 kHz pingers were avoided by Hector's dolphins (Stone et al. 1997) was analytically flawed and hence its conclusion is not correct (Dawson & Lusseau 2005). While setting mortality limits is an effective solution in some fisheries, it requires sufficient observer coverage to provide credible data on how many dolphins are caught, and hence when the fishery should be closed. Baird & Bradford (2000), who analysed the data from the Canterbury observer programme, estimated that the level of observer coverage would need to be 56–83% (depending on the fisheries area) to achieve a CV of 30% on the capture estimate, and 74–100% to achieve a CV of less than 20%. The third solution, creation of spatial closures where harmful activities are restricted or regulated, is the only management approach for which there has been an apparent associated improvement in a vital rate for Hector's and Māui dolphins. Gormley et al. (2012) estimated a 90% probability of increased annual survival rate following the designation of the Banks Peninsula Marine Mammal Sanctuary (see below).

The first spatial closure implemented to mitigate the risk of Hector's dolphin incidental capture was designated at Banks Peninsula in 1988 (Dawson & Slooten 1993). Commercial set netting was effectively prohibited out to 4 nm from the coast and recreational set netting was subject to seasonal restrictions (Dawson & Slooten 1993). A second was designated off the WCNI in 2003. All set nets were prohibited to 4 nm offshore (DOC & MFish 2007). In 2008, a more extensive package of spatial closures was implemented by the Minister of Fisheries (see review by Slooten 2013), providing some protection in most of the areas where Hector's and Māui are found and largely superseding the two existing discrete closures. The set-net restrictions on the WCNI 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) (Figure 6.8), most set netting was prohibited within 4 nm of the coast on the ECSI and SCSI (Figure 6.9 and Figure 6.10), and recreational set netting was banned on the WCSI within 2 nm of the coast and commercial set netting was subject to a seasonal restriction (Figure 6.8). Trawling was banned on the WCNI to 2 nm offshore between Maunganui Bluff and Pariokariwa Point and 4 nm offshore between Manukau Harbour and Port Waikato, and restricted within 2 nm offshore on the ECSI and SCSI[16] (Figure 6.11). 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 and requiring an MPI observer on any commercial set net vessel operating between 2 and 7 nm. 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 (Figure 6.8).

[16] 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.

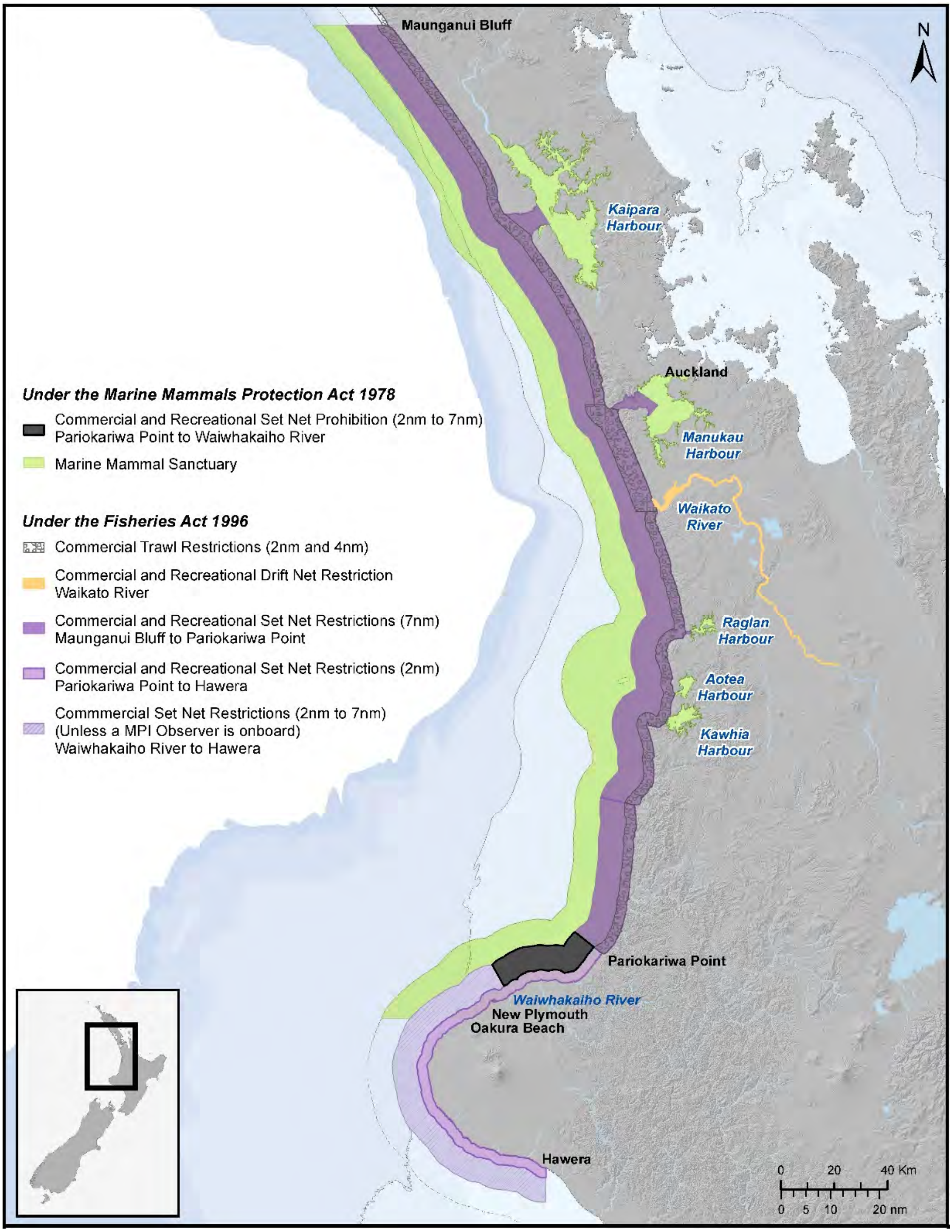

Figure 6.8: Summary of restrictions on commercial and amateur set netting on the WCSI. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

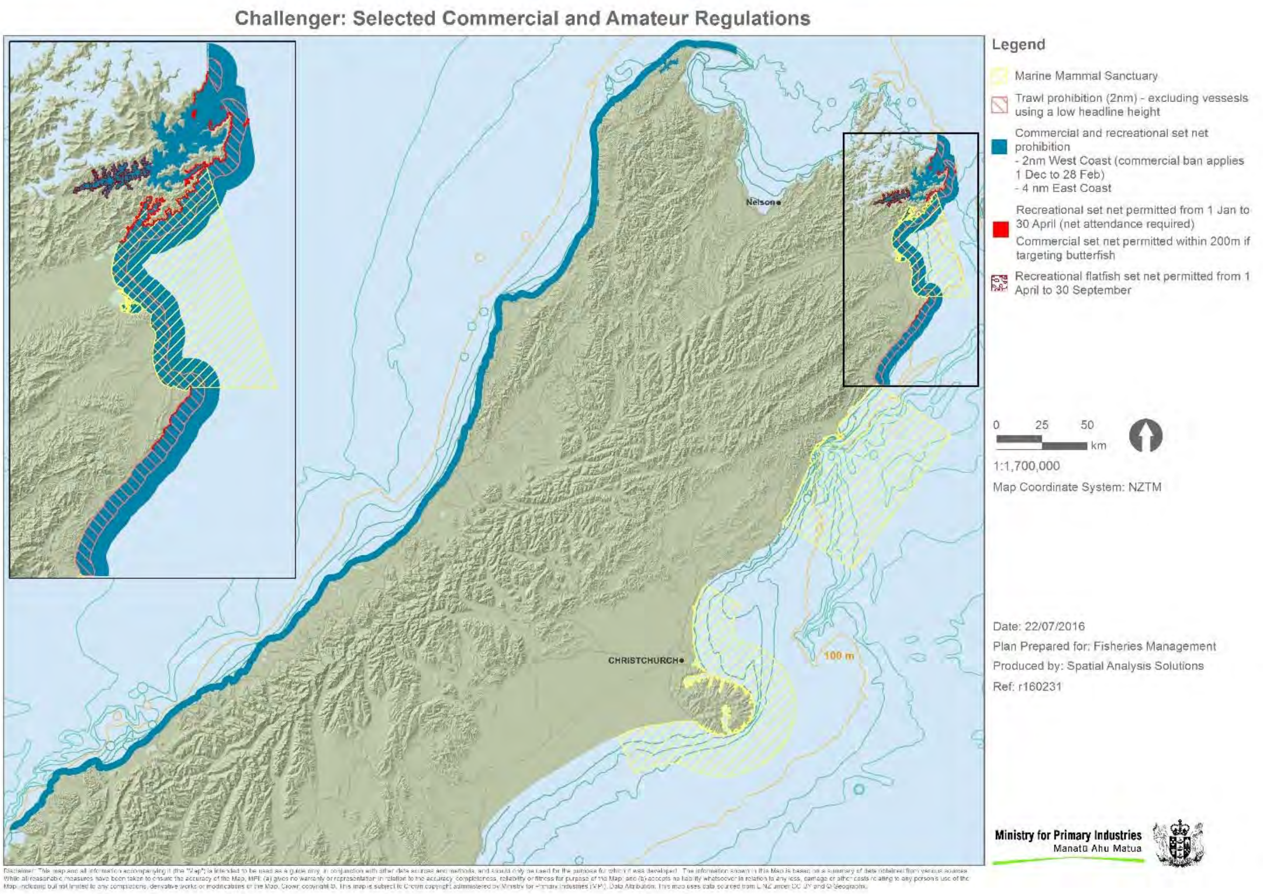

Figure 6.9: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the Challenger area, west coast and northern east of New Zealand. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector’s and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

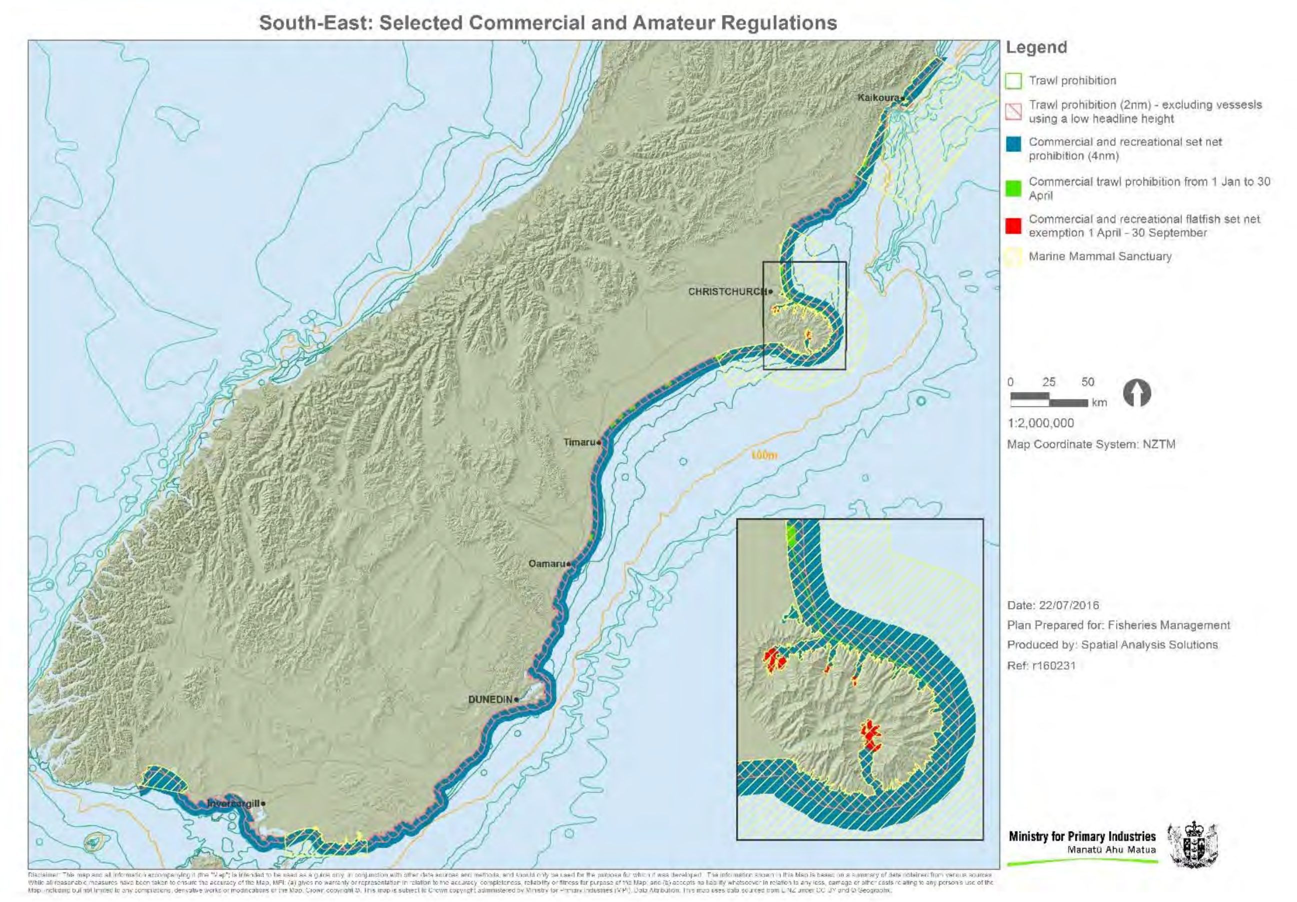

Figure 6.10: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the south-east of New Zealand. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

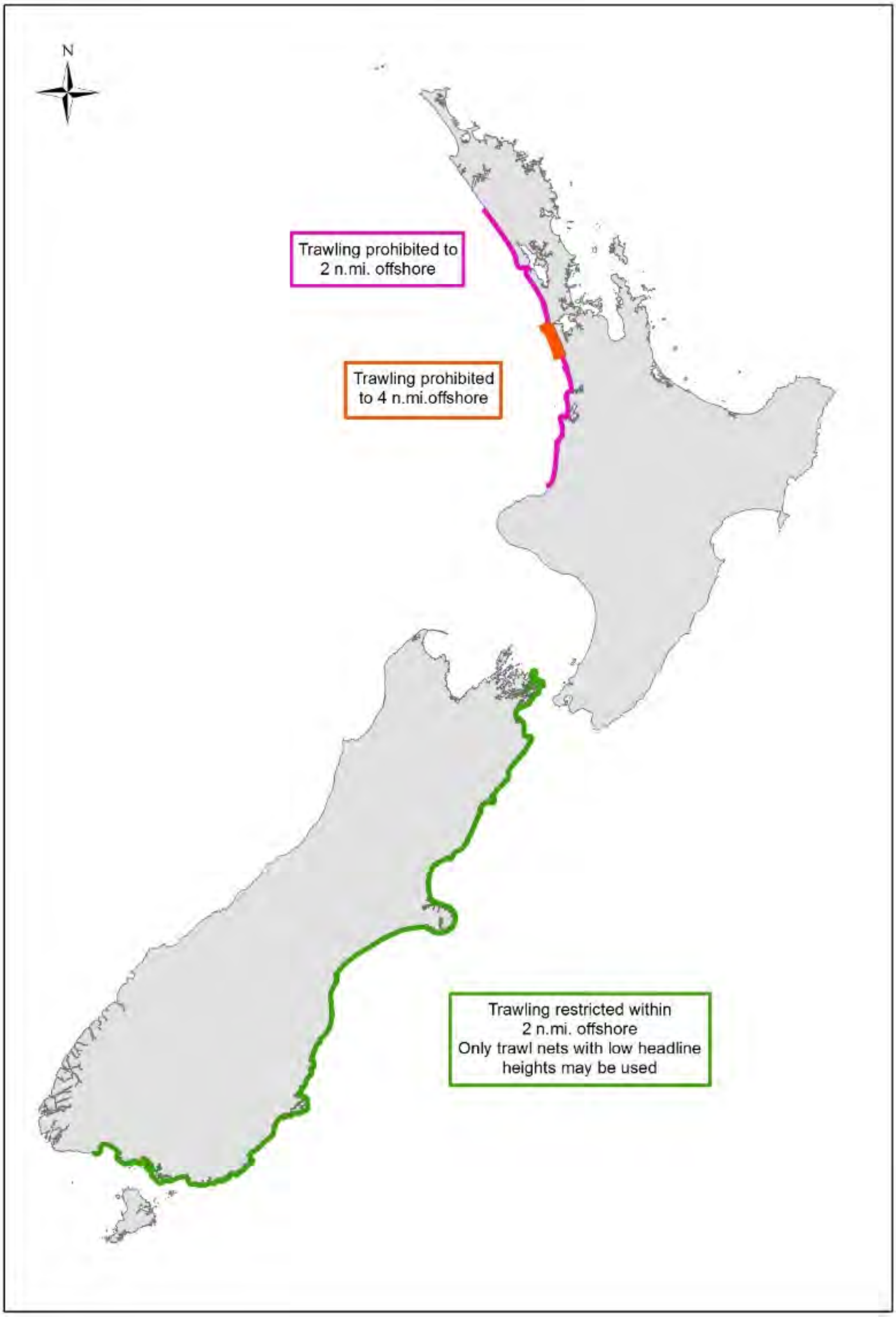

Figure 6.11: Summary of restrictions on trawling. For a full description of the restrictions see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

Assessing the degree of coverage of Hector's and Māui dolphin distribution afforded by spatial management measures is not straightforward as dolphin distributions are dynamic. Aerial surveys can be used to provide a broad-scale indication of dolphin distribution; however they only provide a static picture, strictly relevant to the time of the survey. Notwithstanding this limitation, it is possible to gain an indication of the proportion of a population that was within or outside a particular area at the time of an aerial survey from the proportion of on-effort sightings that were made inside or outside the area. For example, Rayment et al. (2010; Figure 6.12) 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 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 recent ECSI aerial surveys (MacKenzie & Clement 2014; Figure 6.13). In the Banks Peninsula (BP) stratum, 45% of the local summer population and 26% of the local winter population were within the set-net fisheries restriction zones. In the Clifford and Cloudy Bay (CCB) stratum, 47% of the local summer population and 14% of the local winter population were within the set-net fisheries restriction zones Although a sizeable proportion of the sightings occurred within areas closed to set-net fishing during both surveys (Rayment et al. 2010, MacKenzie & Clement 2014), many sightings in summer and most sightings in winter occurred outside these areas.

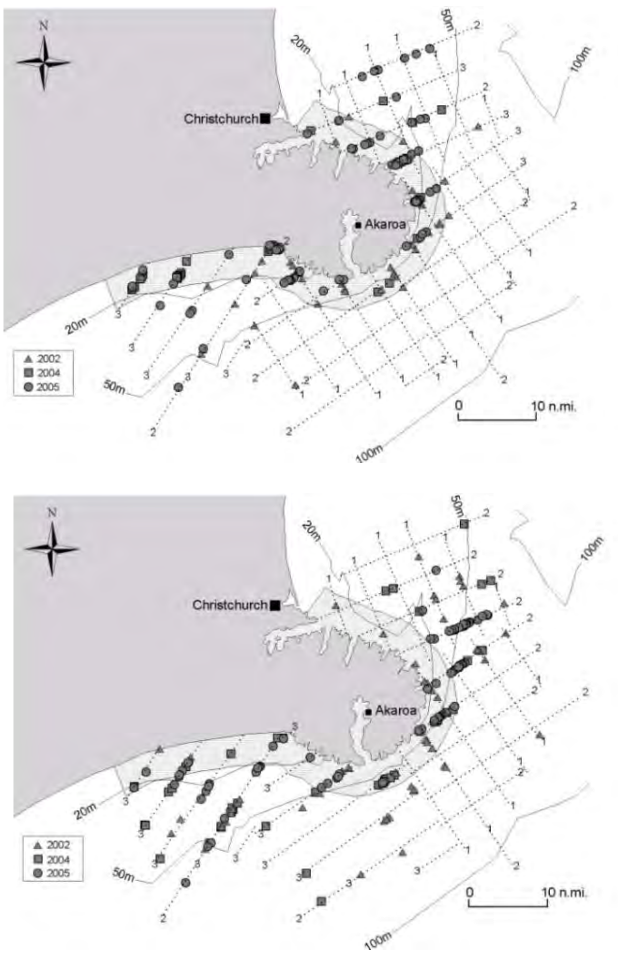

Figure 6.12: Transects and Hector's dolphin sightings on (top) three summer surveys, and (bottom) three winter surveys around Banks Peninsula. Numbers at the end of transect lines are the number of years each line was surveyed. Reproduced from Rayment et al. (2010).

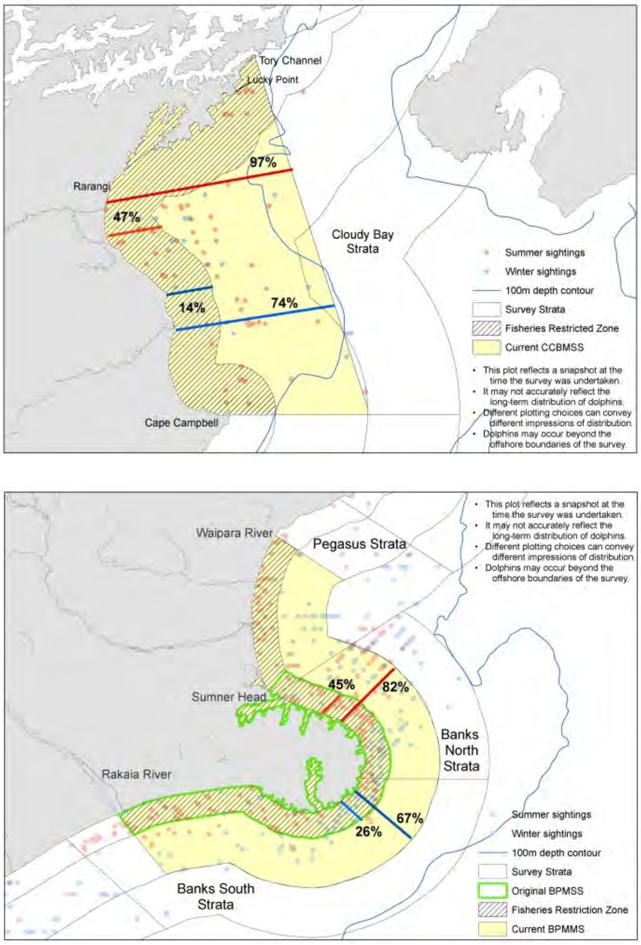

Figure 6.13: The location of summer (red) and winter (blue) survey sightings in relation to fisheries restriction zones and marine mammal sanctuary (MMS) boundaries around Clifford and Cloudy Bays (CCB, top) and Banks Peninsula (BP, bottom). Lines and associated percentages represent proportion of the local population found within 4 nm and 12 nm in summer (red) and winter (blue). Reproduced from MacKenzie & Clement (2014).

6.4.3 PREVIOUS MODELS OF POPULATION-LEVEL FISHERIES IMPACTS

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. Most of this work has been published in science journals (Martien et al. 1999, Burkhart & Slooten 2003, Slooten 2007a, Slooten & Dawson 2010) using their respective peer-review processes, but Davies et al.'s (2008) analysis was reviewed by the AEWG and published as a research report.

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 survey 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 patterns to estimate fishing-related dolphin mortalities for all four populations.

Martien et al. (1999) employed a simple logistic ('Schaefer') population model and projected numbers back to 1970, and forward 200 years, from the 1985 abundance estimate published by Dawson & Slooten (1988). Three separate populations were modelled (WCNI, WCSI and a population that included both ECSI and SCSI populations). Using Dawson's (1991) estimates of mortality from the ECSI area, the back calculation suggested a total of 7077 dolphins across the three populations in 1970, if maximum population growth rate was 4.4%, and 7957 if maximum population growth rate was 1.8% per annum. Martien et al. (1999) considered that the 1985 estimate of abundance was likely to be a slight underestimate (because transects to assess offshore distribution extended only 5 nm offshore), but suggested that any resulting bias in the estimate of the level of the population as a proportion of carrying capacity was likely to be small. The ESCI population was projected to increase for all combinations of parameters except when the maximum growth rate was set to 1.8%.

Davies & Gilbert (2003) conducted a risk assessment for Māui dolphins using a spatially and temporally stratified, age-structured, Bayesian population model for ECSI Hector's dolphins, a population thought to have similar biological and productivity characteristics to Māui dolphin. Estimated population productivity was highly uncertain and largely driven by the priors. Strong assumptions were needed to translate the ECSI model to a model for Māui dolphin and to model population distribution and abundance off the WCNI. Davies & Gilbert found the probability of population decline to be high (50 to 90%) assuming the distribution and intensity of fishing effort pertaining at the time, but the predicted performance of alternative management strategies was sensitive to assumptions about movement, adult survival rate, and set-net catchability. In February 2003 the Ministry of Fisheries introduced closures off WCNI to reduce the risk to Māui dolphins.

Burkhart & Slooten (2003) developed a stochastic version of the logistic model to include a wider range of parameters, variation in fishing effort and population growth, and smaller population units (16 closed populations). Using the same survey and mortality estimates as Martien et al. (1999) yielded similar estimates of the total 1970 population size, but disaggregation of the population into smaller units allowed a conclusion that only the Banks Peninsula subpopulation was likely to increase.

Slooten (2007a) used the stochastic version of the logistic model, the 1998–2003 series of abundance estimates, and catch rates from a 1998 observer programme and concluded a markedly higher estimate of 29 316 individuals in 1970 (CV = 0.26). Slooten's (2007a) projections under status quo management suggested that populations in many areas, including Banks Peninsula, would decrease, but that the WCNI population would increase. Middleton et al. (2007) criticised the high level of confidence ascribed by Slooten (2007a) to her model results without acknowledging that (i) these were dependent on particular model assumptions and (ii) failed to consider other relevant data. In response, Slooten (2007b) gave more detail of her modelling choices, suggested that they were unlikely to lead to overestimation of the impact of fishing, and pointed to similarities between her results and those of other work that was close to being finalised at the time (Davies et al. 2008).

The modelling conducted by Davies et al. (2008) built on the work by Davies & Gilbert (2003) and comprised a Bayesian age-structured population model for the Banks Peninsula subpopulation and 100-year projection simulations for all four subpopulations under different assumed management regimes. The BP population model was structured by age, area, and seasonally to account for the behaviour of the dolphins and the fishery, had a density-dependent calving rate (maximum one calf per female every two years). It was fitted to an absolute abundance estimate from the 1998–2000 surveys of the ECSI, a time series of relative abundance indices for 1990 to 1996 from mark-recapture analyses of dolphin re-sightings around Banks Peninsula, an estimate of average annual adult survival rate 1985–2002, information on the age at first reproduction, the age composition of entangled dolphins, the catch of dolphins recorded by relevant observers, and the amount and distribution of relevant commercial set-net fishing since 1970. Sensitivity to key assumptions was explored by fitting models based on alternative assumptions and by omitting some datasets.

Because so few data were available on the dolphin population and bycatch, Davies et al. (2008) required informative priors to fit their Banks Peninsula model. Even so, the posterior distributions of most parameters were broad and were sensitive to key assumptions, suggesting great uncertainty in our understanding of historical dolphin population dynamics and current population status. Estimates of potential population growth rate ranged from close to zero to the upper bound of what is biologically feasible. The stochastic 100-year projections for each subpopulation entail additional uncertainty, only some of which could be captured in the simulations.

The AEWG agreed that:

- The outcomes of different management strategies could not be predicted with any certainty and, for all subpopulations and management strategies modelled, future population increases and decreases were both plausible.

- Taking the modelling results at face value, all three subpopulations of Hector's dolphin were more likely to decline than increase under set-net fishing effort pertaining at the time, and the decline could be substantial. Conversely, under all alternative strategies simulated, all three subpopulations of Hector's dolphins were more likely to increase than decrease.
- The results for ECSI, including Banks Peninsula, were likely to be more reliable.
- The predicted rates of increase or decrease of all subpopulations were sensitive to the assumed level of productivity.
- For Māui dolphins, the management regime at that time included substantial protection, and the likelihood of continued decline depended strongly on the assumed level of productivity.
- The available data had been used in the best possible way and had been found not to be sufficient to support a definitive analysis. However, the modelling provided helpful guidance on areas where new information should be collected to reduce our uncertainty.
- If the risk analysis was to be communicated to managers, it should be with appropriate caveats around its shortcomings and uncertainty.

The AEWG could not agree whether it was reasonable to adopt all the assumptions required but, consistent with the Terms of Reference, the Chair of the AEWG decided that the modelling could provide qualitative guidance to managers as a risk assessment. He added that the predicted rates of change for all Hector's and Māui subpopulations were sensitive to the assumed level of productivity but, except at the lowest level of productivity, the differences between the predicted outcomes of strategies other than status quo were modest. He noted that, at the lowest assumed level of productivity, projections suggested that the small SCSI subpopulation was more likely to decrease than increase under all simulated management measures other than zero fishing mortality, and that population was also quite likely to be affected by depensation (increasingly low population productivity as abundance decreases, also called an Allee effect).

The stochastic logistic model was used by Slooten & Dawson (2010) to assess the effect of management options developed for the Hector's and Māui Dolphin Threat Management Plan (although the options evaluated differed from the final proposals). The input data were similar to those of Slooten (2007a, 2007b). Slooten & Dawson's (2010) population estimates for 1970 (their figure 1) were similar to those reported by Slooten (2007a), but showed some regional differences. Both Slooten (2007a) and Slooten & Dawson (2010) suggested that the WCNI population would increase under management pertaining at the time, whereas the other three populations would decline. Slooten & Dawson (2010) further suggested that their option B (similar to the 2008 measures) would lead to the ECSI and SCSI populations increasing on average, whereas the WCSI population would continue to decline.

Slooten & Davies (2012) published a new estimate of 23 captures from the ECSI population between May 2009 and April 2010 based on observer records (although their description of the methods suggests that their reported CV of 21% is greatly underestimated). They used this and an estimate of 110–150 dolphins caught annually around the South Island before 2008, including 35 to 50 dolphins caught off the ECSI (Davies et al. 2008) to update the two most recent modelling approaches (Davies et al. 2008, Slooten & Dawson 2010). Slooten & Davies (2012) found the consistent predictions from all population models used to date surprising, given the substantial differences in their structural assumptions. They noted that all population models indicated that substantial declines had occurred and were likely to continue, and concluded that this consistency should add confidence to the predictions about the consequences of the different management options. In addition, they also cited a number of reasons why the conclusions might be optimistic, notably that most only include incidental captures in commercial set nets, as the other forms of fisheries-related mortality have yet to be quantified (Davies et al. 2008, Slooten & Dawson 2010, Slooten & Davies 2012).

6.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

At the time of the Māui dolphin risk assessment described in Currey et al. (2012), fishery threats were only estimated subjectively, because comprehensive spatial distributions had not yet been estimated for Māui and Hector's dolphins, and fisheries observer coverage was low, especially for set-net fisheries. However with the subsequent availability of aerially or expert derived spatial distributions (Figures 6.1 and 6.6) and development of the SEFRA method (Chapter

3) fisheries risk can now be estimated empirically based on spatial overlap between dolphins and fishing effort, and observed capture rates.

The first implementation of the New Zealand Marine Mammal Risk Assessment (MMRA) is described in Abraham et al. (2017). The MMRA estimates that Māui and Hector's dolphins are the third and fourth most at-risk subspecies of marine mammal, respectively, from New Zealand commercial fisheries (Figure 6.14). For Māui dolphin the mean estimated risk ratio is 0.47 (95% c.i.: 0.00–1.33). For Hector's dolphin the estimated mean risk ratio is 0.45 (95% c.i.: 0.18–0.92).[17]

Fisheries risk to Māui and Hector's dolphins is attributable primarily to set-net fisheries, and secondarily to inshore trawl (Figure 6.14). For Hector's dolphins, the mean estimate is of 9.0 fatalities per year in trawl fisheries (95% c.i.: 1.1–26.6) and 32.3 fatalities per year in set-net fisheries (95% c.i.: 13.8–65.8). For Māui dolphins, the mean estimate is of 0 (0–0.1) fatalities per year in trawl fisheries, and 0.2 (0.0–0.5) fatalities per year in set-net fisheries. These numbers include estimated observable captures and an allowance for cryptic mortality.

That estimated fishery deaths are considerably lower than previous estimates likely reflects: i) the improved methods employed, allowing empirical estimation with explicit consideration of spatially unrepresentative observer coverage; ii) improved data inputs with respect to population size and species spatial distribution, arising from South Island aerial surveys; and iii) changes in actual fisheries risk since the adoption of spatial fisheries closures in areas of highest dolphin density (noting that subjective expert estimates from Currey et al. (2012) likely also reflected risk in areas that are now closed to fishing).

The following sections are reproduced from Abraham et al. (2017).

6.4.4.1 HECTOR'S DOLPHIN

The estimated annual potential fatalities of Hector's dolphin were 41.3 (95% c.i.: 19.1–77.7), with 32.3 (95% c.i.: 13.8–65.8) estimated captures in set-net fisheries. The Hector's dolphin subspecies has three genetically distinct subpopulations: East Coast South Island, West Coast South Island, and South Coast South Island (Baker et al. 2002). The overlap between set-net fisheries and Hector's dolphin was almost entirely on the East Coast South Island. The current estimate of captures in set-net fisheries overlaps with a previous estimate of 23 Hector's dolphin (CV: 0.21) caught in east coast South Island set-net fisheries during 2009–10 (Slooten & Davies 2012). The risk to Hector's dolphin was entirely less than one, however, there was a 9.4% probability that the risk to the east coast South Island Hector's dolphin population from set-net fishing in that area exceeded one.

There are already extensive areas of South Island waters where set-net fishing is prohibited, including restrictions within 4 nm (7.4 km) off the coast, and a larger marine mammal protection area surrounding Banks Peninsula. The effect of these restrictions is evident on the map of overlap between Hector's dolphin and set-net fisheries, as there is no overlap close to shore along most of the South Island coast. The set-net closures are reflected in the fishing effort used for the risk assessment (from the period 2012–13 to 2014–15).

For Hector's dolphin, the uncertainty in the risk was high. This uncertainty were partly due to the low observer coverage of set-net fisheries: around half of the observed captures were from a dedicated programme that was carried out in the late 1990s (Baird & Bradford 2000, Starr & Langley 2000). Trials of video monitoring on set-net vessels have demonstrated that Hector's dolphin bycatch can be recorded by video cameras (e.g., McElderry et al. 2007), and it appeared that the video was able to record captures that would not have been seen by observers on the vessels. Expanding observer coverage (either via human observers or video monitoring) would help to reduce uncertainty in the estimated captures. In addition,

[17] These risk ratios reflect a Population Sustainability Threshold in the risk ratio formula applying a default population reference outcome for the multi-species risk assessment, i.e., 'recovery to or stabilisation at a level higher than 50% of carrying capacity, with 90% certainty'. Note however that in other species-specific implementations of the SEFRA method, alternative population reference outcomes may be adopted, reflecting species-specific policy choices.

improving knowledge of cryptic mortality would reduce uncertainty, as estimated annual potential fatalities are in part associated with an assumed cryptic mortality. For set-net fisheries, the probability that a capture incident is observed was assumed to be uniformly distributed between one-third and one, with a mean of two-thirds. Obtaining better information on the drop-out rates, i.e., the proportion of dolphin that are caught but not recovered onboard the vessel, would help reduce the uncertainty. Another source of uncertainty in the estimation of risk to Hector's dolphin was the estimation of the maximum growth rate (r_{max}; mean 0.026; 95% c.i.: 0.018–0.036), which relied on expert judgement. Research to estimate r_{max} empirically from demographic data or life-history parameters may be useful to reduce this uncertainty. The analysis of overlap relied on surveys conducted during two seasons (winter and summer), and no information was available on the variation in the distribution of Hector's dolphin within those seasons.

From the observed set-net captures, the model estimated a live-release probability of 12.8% (95% c.i.: 3.7–26.4%) for small dolphins in set-net fisheries. The post-release survival of these animals was unknown, and so was assumed to be uniformly distributed between zero and one, with a mean of one-half. Taken together, around 6% of the capture incidents were assumed to not have resulted in a fatality.

6.4.4.2 MĀUI DOLPHIN

The mean and median values of the risk ratio for Māui dolphin were below 0.5, but the upper credible limit extended above one. Set-net fisheries were the only fisheries with a mean of more than 0.05 annual potential fatalities of Māui dolphin, highlighting that efforts to reduce the potential capture of this species in fisheries need to focus on set-net fishing.

The estimated overlap between set-net fisheries and Māui dolphin was concentrated inside harbours on the North Island west coast. In the region of overlap, much of the fishing effort had the location imputed. To help refine the overlap, and consequently the estimated captures, it is necessary to clarify where fishing effort in these harbours is occurring, and how the distribution of Māui dolphin extends into them. Uncertainty in estimates of risk to Māui dolphin also reflected uncertainty in the maximum growth rate, r_{max}, which was estimated by experts as mean 0.023 (95% c.i.: 0.015–0.034).

Based on the assumed distribution of Māui dolphin, the risk assessment suggests that potential fatalities of Māui dolphin would be reduced by extending the set-net ban into North Island west coast harbours, particularly Kaipara, Raglan, Aotea, and Kāwhia harbours. There was also overlap with set-net fisheries operating near New Plymouth, toward the south of the range of Māui dolphin. Conclusions from the current study are sensitive to assumptions about the distributions of Māui dolphin and of unlocated set-net effort within harbours. The vulnerability of Māui dolphin was assumed to be the same as for Hector's dolphin (they were treated as the same species in the model). The estimated capture rate largely depended on observations made on the South Island east coast of Hector's dolphin. Since 2012–13, there has been observer coverage of set-net fisheries in the Taranaki area, focused on the warehou set-net fishery that operates near New Plymouth. Between 2012–13 and 2014–15, observer coverage of the minor species set-net fishery (which includes warehou targets) in the Taranaki region has varied between 38% and 73%. There were no observed captures of Māui dolphin.

6.4.5 ONGOING AND SPECIES SPECIFIC RISK ASSESSMENT

The MMRA was based on the SEFRA method described in Chapter 3, but included some modifications with particular relevance for Hector's and Māui dolphins. For example the model in Abraham et al. (2017) included both a 'species group' variable (in which Hector's and Māui dolphins were grouped with other small dolphins) and a species-specific variable (with Hector's and Māui dolphins recognised as a single species). Crucially, concerns raised by the expert review of Lonergan et al. (2017) about the reliance on Delphi-derived species spatial distributions were not applicable to Māui and Hector's dolphin specifically, because the MMRA instead utilised the distributions shown in Figures 6.1 and 6.6. However the 'small dolphin' species group variable also included other species such as common dolphin for which empirically derived distributions were not available. To the extent that the species group variable affects the estimated vulnerability of Hector's and Māui dolphins, captures and risk estimates for these species may still reflect Delphi-derived distributions. These estimates will be updated when new cetacean distributions are available (from PRO2014-01).

For Māui dolphins in particular, the current MMRA risk estimates remain sensitive to assumptions about to what extent the dolphin distribution extends offshore and into the mouths of North Island harbours. Research is underway under project PRO2017-12 to refine the estimated spatial distribution in Figure 6.1 using spatial modelling informed by acoustic detections and sightings data. Māui dolphin risk estimates are also sensitive to the spatial distribution of small vessel set-net fishing effort in harbours (which historically has only been reported within Statistical Areas, not with precise locations). As positional effort reporting improves, the risk associated with harbour set-net fishing can be estimated with higher precision.

The current MMRA treats risk to Hector's dolphins as a single species, but overlap with fisheries varies between subpopulations. Separately, under project SEA2016-30, MPI is developing the capacity to disaggregate and query spatial risk assessment outputs on a regional basis for Hector's and Māui dolphins, and to examine sensitivities

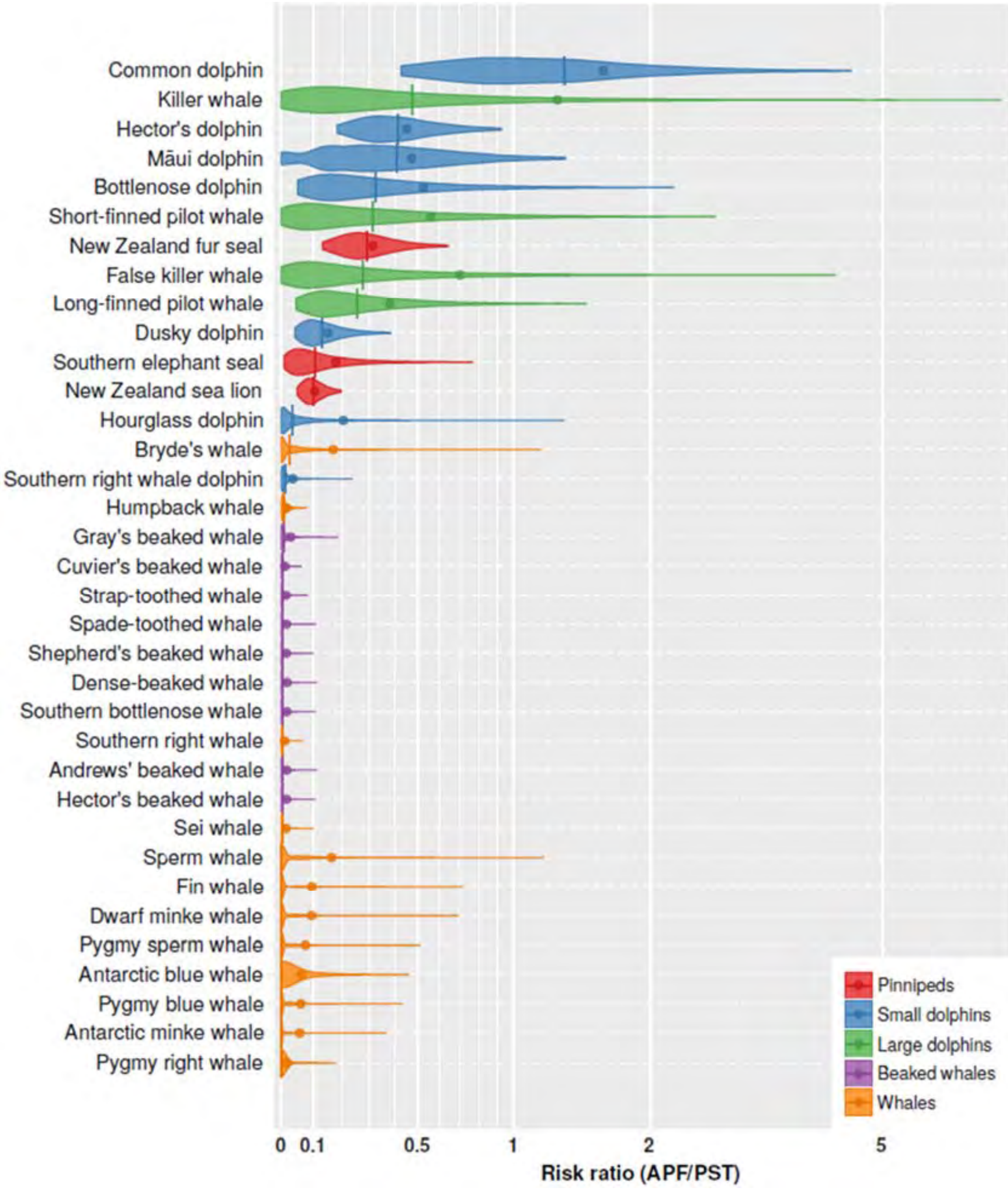

Figure 6.14: Risk ratio for New Zealand marine mammals, calculated as the ratio of fishery-related deaths (APF) to the Population Sustainability Threshold (PST). Values are displayed on a logarithmic scale, and the distribution of the risk ratios within their 95% credible interval indicated by the coloured shapes, including the median risk ratio (vertical line). Species are listed in decreasing order of the median risk ratio.

(e.g., alternative spatial distributions of dolphins or of fishing effort). Outputs of this project will allow managers to consider the consequences of alternative risk management scenarios, to inform the update of the TMP in 2018.

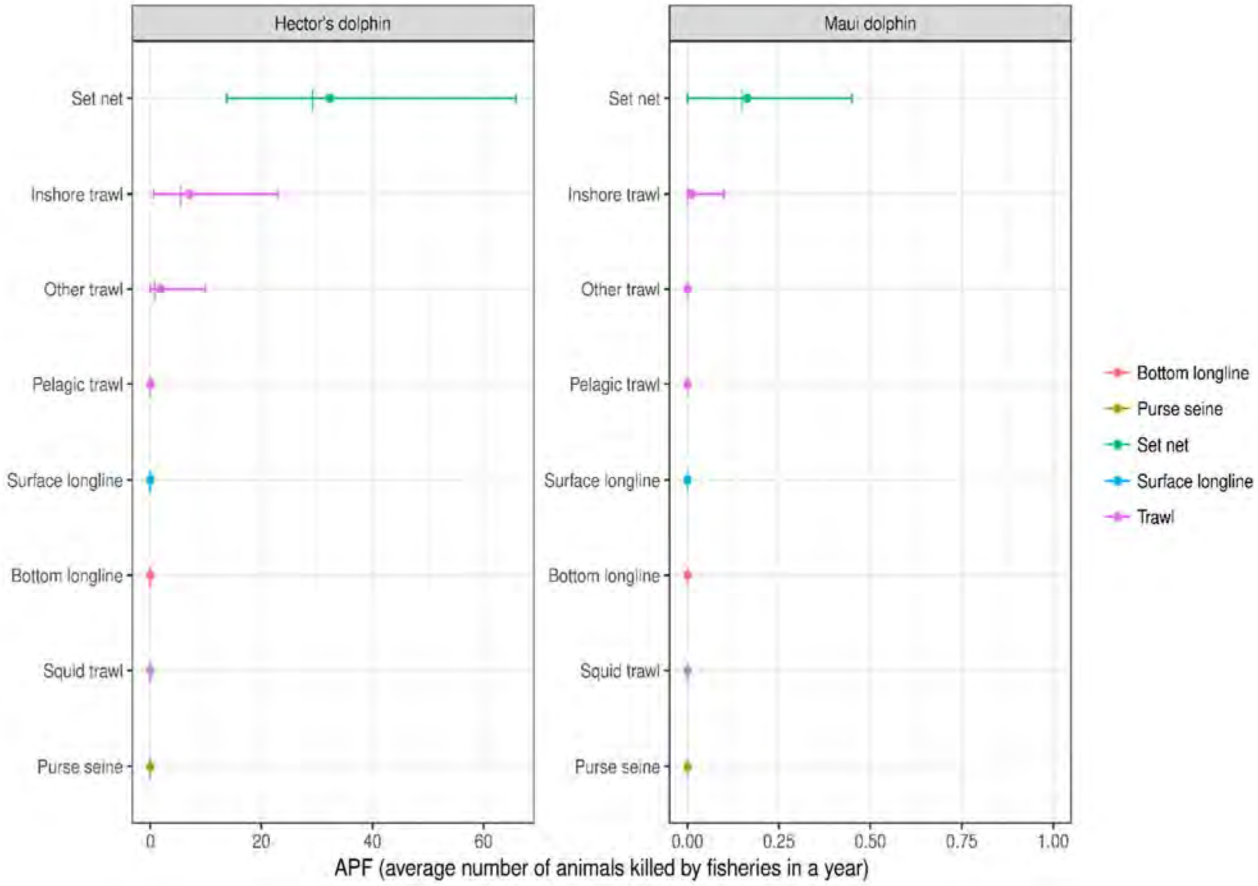

Figure 6.15: Estimated annual fishery-related deaths of Hector's dolphins and Māui dolphins by fishery group, from the multi-species New Zealand Marine Mammal Risk Assessment (Abraham et al. 2017).

6.4.6 SOURCES OF UNCERTAINTY

The uncertainties and assumptions in the modelling by Davies et al. (2008), Slooten & Dawson (2010), and Slooten & Davies (2012) were reviewed in detail by Slooten & Davies (2012). The models incorporate uncertainties in parameter distributions and hence population estimates are presented with their estimated levels of precision. The population viability analyses incorporated a distribution for population growth rate based on a wide range of values for maximum growth rate in Hector's dolphin (e.g., Slooten et al. 2000) and the Bayesian population models included a fully integrated parameter estimation of fisheries-related mortality and reproductive rate (Slooten & Davies 2012). Slooten & Dawson (2010) showed via sensitivity analysis that the probability of recovery to half the maximum population size was robust to uncertainty in the catch rate (± 0.25 times the assumed catch rate of 0.037 dolphins per set) used in the PVAs.

The AEWG discussed outstanding areas of uncertainty and concluded that the following areas represented important uncertainties in assessing the impacts of fishing on Hector's and Māui dolphins.

CAPTURE ESTIMATES AND CAPTURE RATE

Increased observer coverage, using a combination of observers and electronic monitoring, for set-net and inshore trawl fisheries is needed to reduce uncertainty in the estimates of captures and capture rate. Observer coverage should cover a sufficiently high proportion of fishing effort so as to enable the detection of rare events (particularly important for Māui dolphin), and to minimise the risk of mistaken risk estimation arising from non-representative coverage.

CRYPTIC MORTALITY

The level of cryptic mortality associated with fisheries interactions is unknown for Hector's and Māui dolphins, but may be non-trivial if estimates for other small cetaceans are any indication (e.g., 58% of captured porpoises falling out of a net before reaching the deck; Kindt-Larsen et al. 2012). Improving our understanding of cryptic mortality is a priority for future risk assessments for Hector's and Māui dolphins.

DEMOGRAPHIC PARAMETERS

All the various risk analyses completed to date rely, at least in part, on demographic data obtained from one part of one population (i.e., Banks Peninsula). This necessitates assumptions as to how these data, and the resulting parameter estimates, were derived and how they apply outside the Banks Peninsula region. Obtaining additional demographic data from other region(s) could enable any difference between regions to be detected and reflected in future risk analyses. However, robust estimation of demographic parameters will require analysis of long-term (more than 10 years) of mark-recapture data collection to produce a time series of photographic or genetic individual identifications. A review of relevant demographic parameters and their derivation will be completed in 2018 as part of project PRO2017-12.

POPULATION ESTIMATES FOR THE SCSI POPULATION

Recent estimates of abundance are available for all populations of Hector's and Māui dolphins (Clement et al. 2011, Hamner et al. 2012b, MacKenzie & Clement 2014 and 2016). The reanalysis of the survey data for the SCSI indicated a notably smaller estimate (median = 238; 95% c.i.: 113–503) than that obtained by Clement et al. (2011) (628; 95% c.i.: 301–1311). An updated abundance estimate for the SCSI population will be obtained under project PRO2016-09.

POPULATION CONNECTIVITY AND MOVEMENT

Ongoing photo-ID research (e.g., Bräger et al. 2002, Rayment et al. 2009b) and genetic recaptures (Oremus et al. 2012, Hamner et al. 2012a, 2012b, 2014) will improve estimates of movements and dispersal (Rayment et al. 2009b, Hamner et al. 2012a, 2012b, Pichler 2002). For example, Hamner et al. (2014) suggested that failure to protect the habitat between the North and South Island will reduce the likelihood of dispersal, possibly to the detriment of Māui dolphin.

SPATIAL DISTRIBUTIONS

Application of the SEFRA method relies upon accurate estimation of species spatial distributions, including seasonal variation (if any). These are available from aerial surveys for South Island Hector's dolphin populations, but due to low population size aerial survey alone was judged to be inadequate to estimate Māui dolphin distributions. Where populations are low and even rare events may produce substantial risk there is a need to estimate risk even in the low-density 'tails' of the spatial distribution where animals occur only rarely. This work will be progressed via spatial distribution modelling under project PRO2017-12.

OTHER THREATS (NON-FISHING-RELATED, INDIRECT, SUB-LETHAL, CUMULATIVE)

Uncertainty exists over the magnitude of impacts faced by Hector's and Māui dolphins due to disease, mining and hydrocarbon extraction, tourism, vessel traffic, anthropogenic noise, pollution, aquaculture and research activities (DOC & MFish 2007, Currey et al. 2012, MPI & DOC 2012). Even if the impacts in isolation are sub-lethal, it is unknown whether the effects are cumulative, how they might affect factors such as breeding success, and whether they interact with the direct and indirect threats due to fishing (DOC & MFish 2007, Currey et al. 2012). Roe et al. (2013) identified infection with *Toxoplasma gondii* as a factor potentially contributing to the population decline of Hector's and Māui dolphins, and recommend further investigation of the source and route of entry of pathogens into the coastal environment. This work is in progress and will be summarised and considered in a spatially explicit manner under project PRO2017-12, to inform the update of the TMP.

6.4.7 POTENTIAL INDIRECT THREATS

Miller et al. (2013) note that red cod is targeted by the inshore trawl fishery and its abundance is highly variable, particularly around Banks Peninsula. Given that red cod contribute most in terms of mass to the diet of Hector's dolphins on the ECSI, Miller et al. (2013) suggest that further research is required to investigate potential trophic effects of fisheries on Hector's dolphin populations.

6.5 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: 238 (s.e.: 94; 95% c.i.: 113–503) in 2011[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 stabilisation from 2010–11 to 2015–16. ECSI Hector's dolphins: Probably declining. Inconsistent evidence from abundance estimates, risk analyses and demographic estimates of population growth rate. WCSI Hector's dolphins: Unknown; ECSI estimates of vulnerability and productivity applied to this area via risk analyses suggest a much lower rate of capture, due to low overlap with fisheries. SCSI Hector's dolphins: Unknown. Inconsistent evidence from abundance estimates and risk analyses.
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 Endangered, Criterion C(1/1), Conservation Dependent in 2013[23] IUCN: Endangered, Criterion A4d in 2013[23]
Number of captures[24]	Māui dolphins: <1 per annum (Davies et al. 2008), 4.97 per annum (95% c.i.: 0.28–8.04; Currey et al. 2012), 0.2 estimated annual potential fatalities (APF) (95% c.i.: 0.00–0.5)[24] ECSI Hector's dolphins: 35 to 50 per annum (Davies et al. 2008) WCSI Hector's dolphins: 70 to 100 per annum (Davies et al. 2008) SCSI Hector's dolphins: about 2 per annum (Davies et al. 2008) All Hector's dolphins: 41.3 estimated annual potential fatalities (APF) (95% c.i.: 19.1–77.7)[24]
Trends in interactions	Possible reduction from 35 to 50 per annum (Davies et al. 2008) to about 23 for ECSI (Slooten & Davies 2012) and 21.6 for all South Island (Abraham et al. 2017).

6.6 REFERENCES

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7 COMMON DOLPHIN (*DELPHINUS DELPHIS DELPHIS*)

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	All of 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.
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)

Note: This chapter has been updated for the 2017 AEBAR with recent capture estimation and risk assessment results.

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).

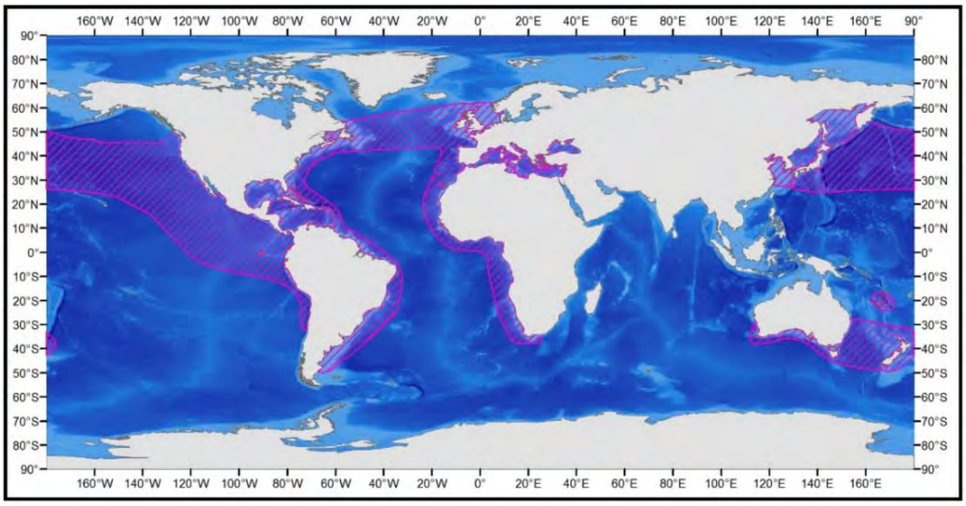


Figure 7.1: Worldwide distribution of short-beaked common dolphins (*Delphinus delphis delphis*) provided by the International Union for the Conservation of Nature (IUCN) (Hammond et al. 2008). Magenta hatched areas indicate range.

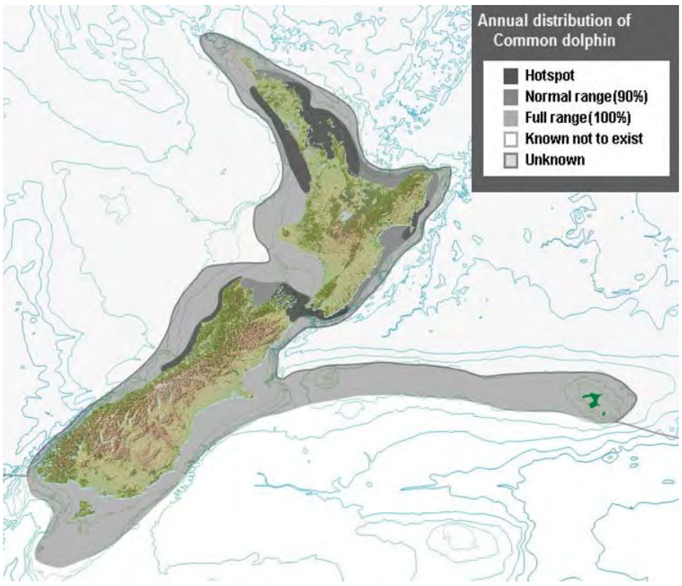


Figure 7.2: Distribution of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters (from www.nabis.govt.nz).

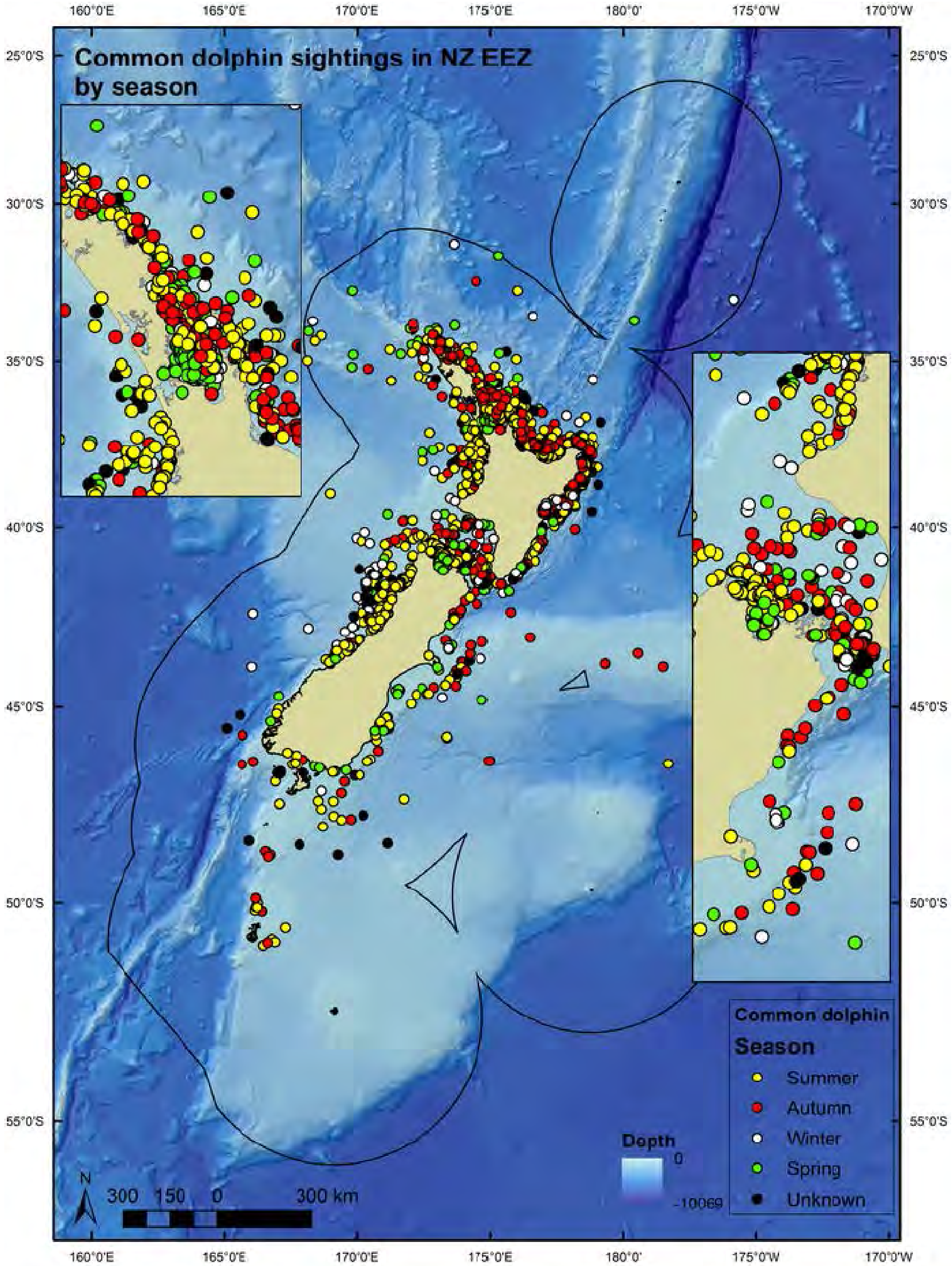


Figure 7.3: Systematic and opportunistic sightings of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters between 1970 and 2013. Data sources include Department of Conservation (DOC), Cawthorn (2009), opportunistic at-sea sightings (NIWA), and the Centralised Observer Database (COD). (Sightings are indicative of the distribution only). Figure from Berkenbusch et al. (2013).

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).

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).

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).

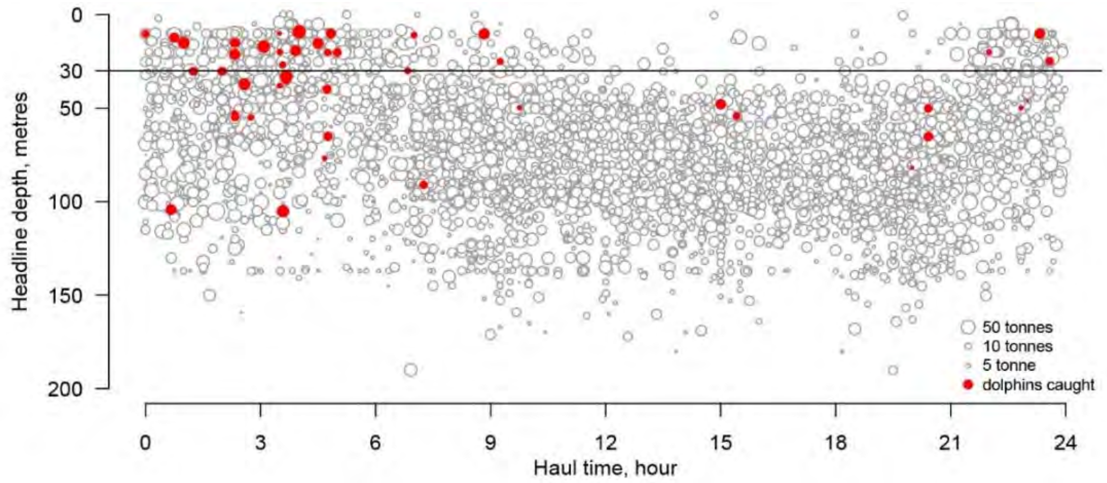


Figure 7.4: Headline depth versus the haul time for observed trawls in the large-vessel jack mackerel fishery. The catch weight is indicated by the size of the circles. Tows where an observed common dolphin capture event occurred are filled (from Thompson et al. 2010).

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

Table 7.1 [Continued]:

Fishing year	Region	Gear	Time of day	Effort		Observed captures	
				Fishing tows	% observed	Mortality	Events
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

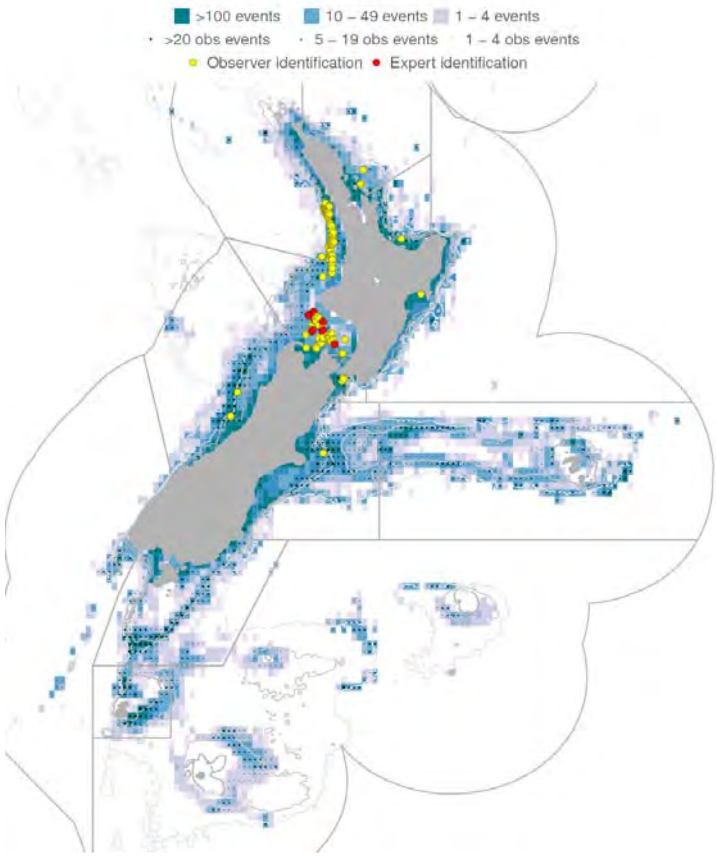


Figure 7.5: Distribution of all trawl fishing effort and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2015–16 (for more information see MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2017001). Fishing effort is mapped into 0.2-degree cells, coloured 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.

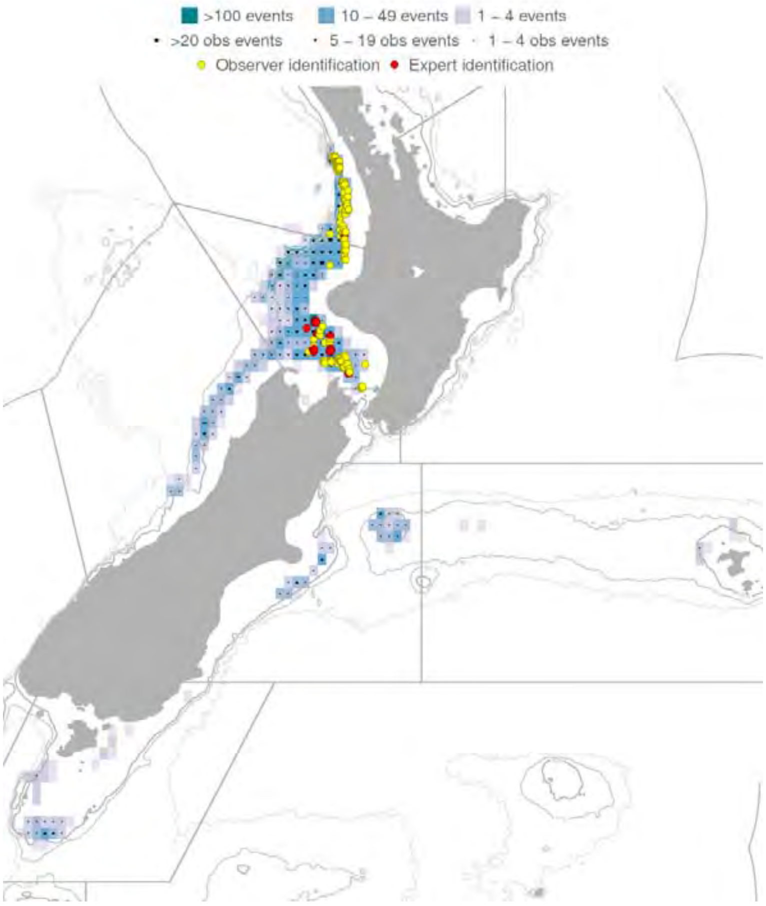


Figure 7.6: Distribution of trawl fishing effort for jack mackerel and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2015–16 (for more information see MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2017001). Fishing effort is mapped into 0.2-degree cells, coloured to represent the number of tows. Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell.

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://data.dragonfly.co.nz/psc, data version 2017001). 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). [Continued on the next page]

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% observed	Number	Rate	Mean	95% c.i.
2002–03	130 175	5.3	21	0.31	171	92–290
2003–04	120 847	5.4	17	0.26	147	82–245
2004–05	120 464	6.4	22	0.29	126	80–185
2005–06	109 951	6.0	4	0.06	45	23–76
2006–07	103 327	7.7	11	0.14	82	45–132
2007–08	89 536	10.1	20	0.22	73	46–107
2008–09	87 550	11.4	20	0.20	64	41–95
2009–10	92 889	10.3	4	0.04	52	26–90
2010–11	86 091	8.8	9	0.12	92	59–153
2011–12	84 424	11.1	5	0.06	34	17–58
2012–13	83 832	14.8	17	0.14	46	30–71

Table 7.2 [Continued]:

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% observed	Number	Rate	Mean	95% c.i.
2013–14	85 113	15.5	30	0.23	58	42–80
2014–15	78 754	17.6	21	0.15	46	32–67
2015–16	78 040	18.2	8	0.06		

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://data.dragonfly.co.nz/psc, data version 2017001). 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	130	55–243
2003–04	2 383	6.4	17	11.18	106	46–197
2004–05	2 510	22.2	21	3.76	82	44–135
2005–06	2 808	25.2	2	0.28	11	2–30
2006–07	2 711	29.6	11	1.37	50	21–95
2007–08	2 652	30.8	20	2.45	43	25–70
2008–09	2 169	37.5	11	1.35	27	13–50
2009–10	2 407	32.7	4	0.51	24	6–55
2010–11	1 882	31.6	7	1.18	63	24–120
2011–12	2 032	76.2	5	0.32	7	5–14
2012–13	2 211	87.7	15	0.77	18	15–23
2013–14	2 448	89.4	28	1.28	31	28–38
2014–15	1752	86.2	19	1.25	23	19–30
2015–16	1 545	89.5	2	0.14		

a) EEZ

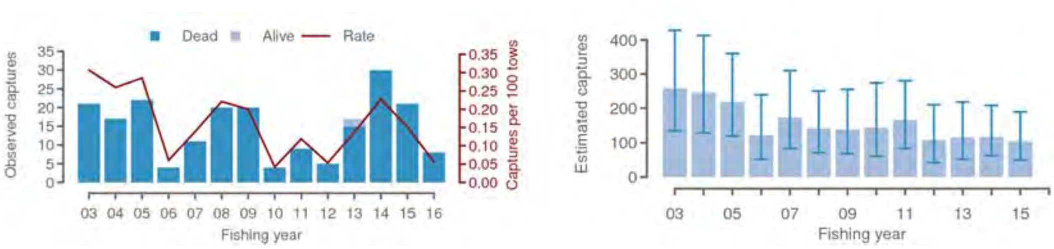
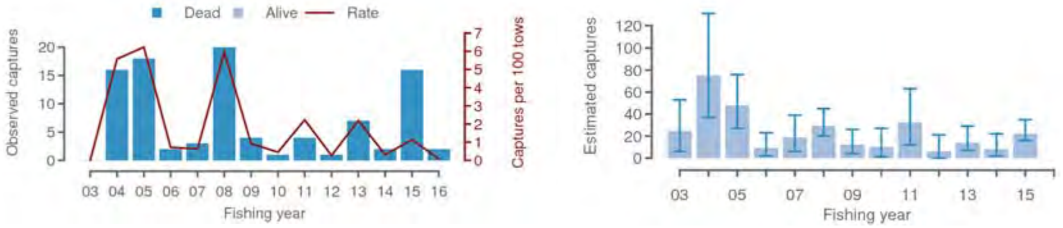


Figure 7.7: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in all trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2015–16, inclusive of three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2017001). 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. [Continued on next page]

b) West Coast North Island



c) Taranaki

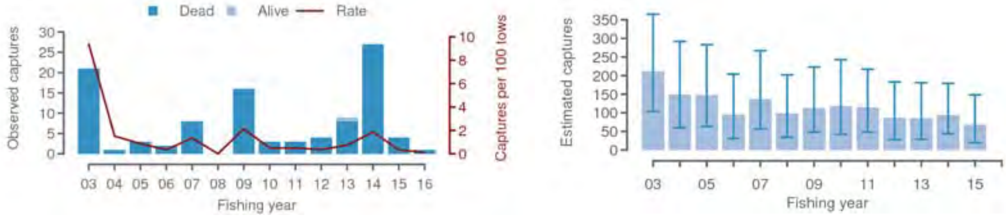
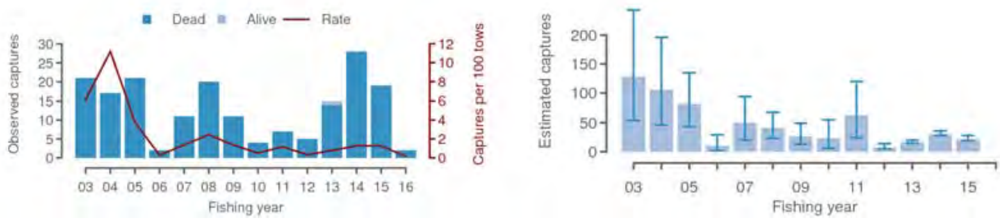
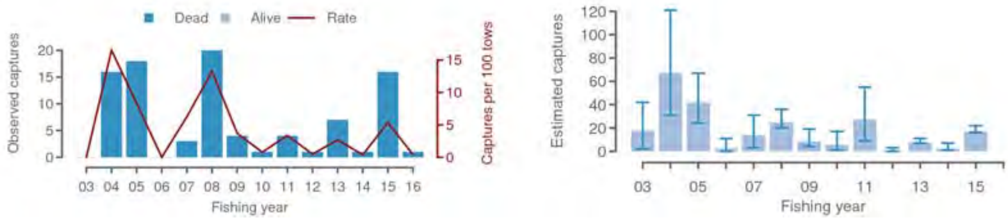


Figure 7.7 [Continued]

a) EEZ



b) West Coast North Island



c) Taranaki

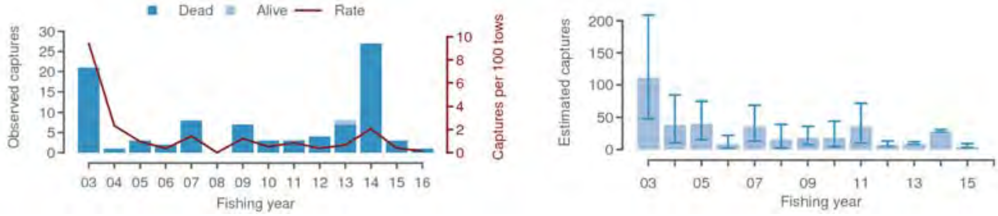


Figure 7.8: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in the jack mackerel trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2015–16 for three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://data.dragonfly.co.nz/psc, data version v2017001). 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.

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).

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.

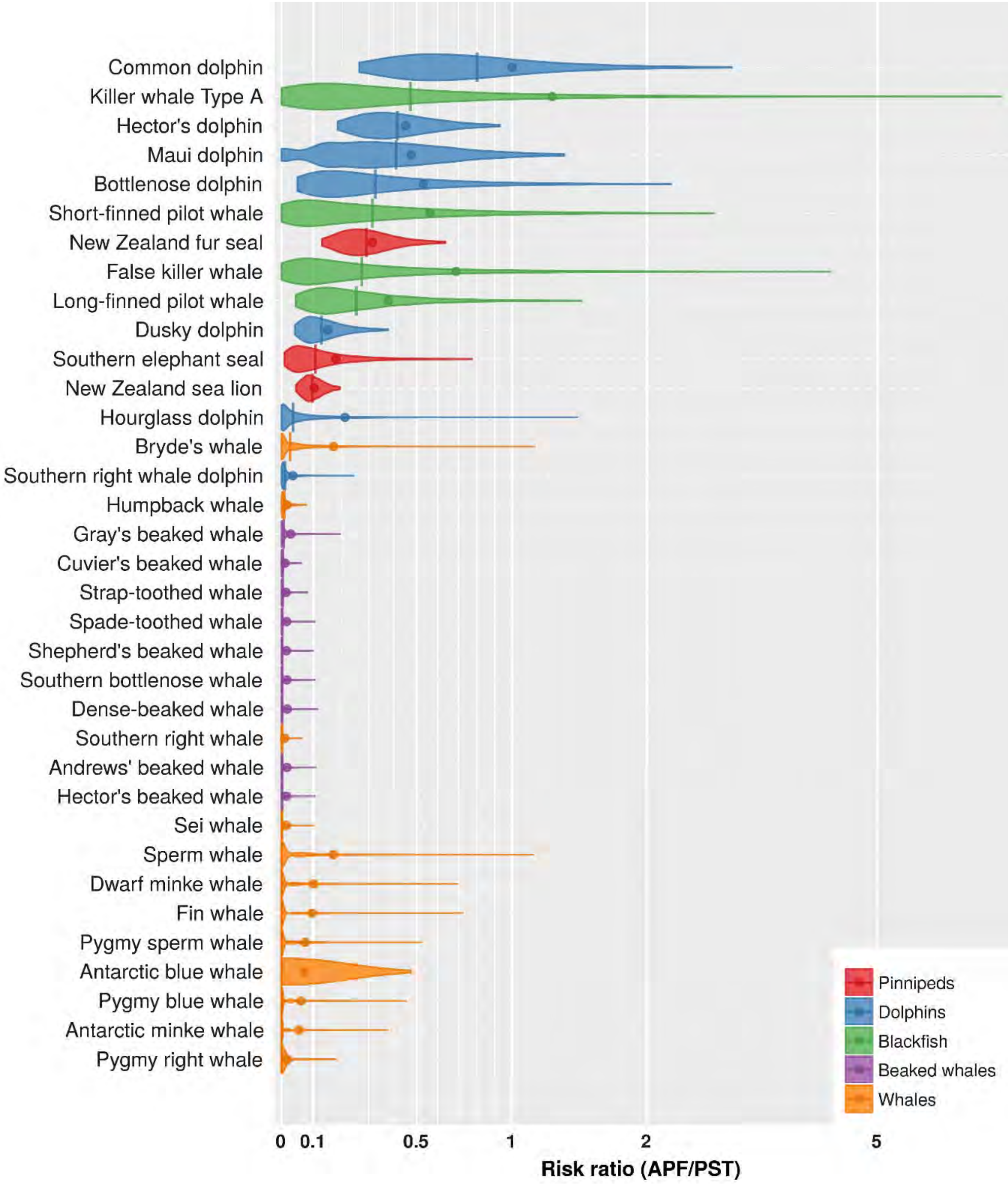


Figure 7.9: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

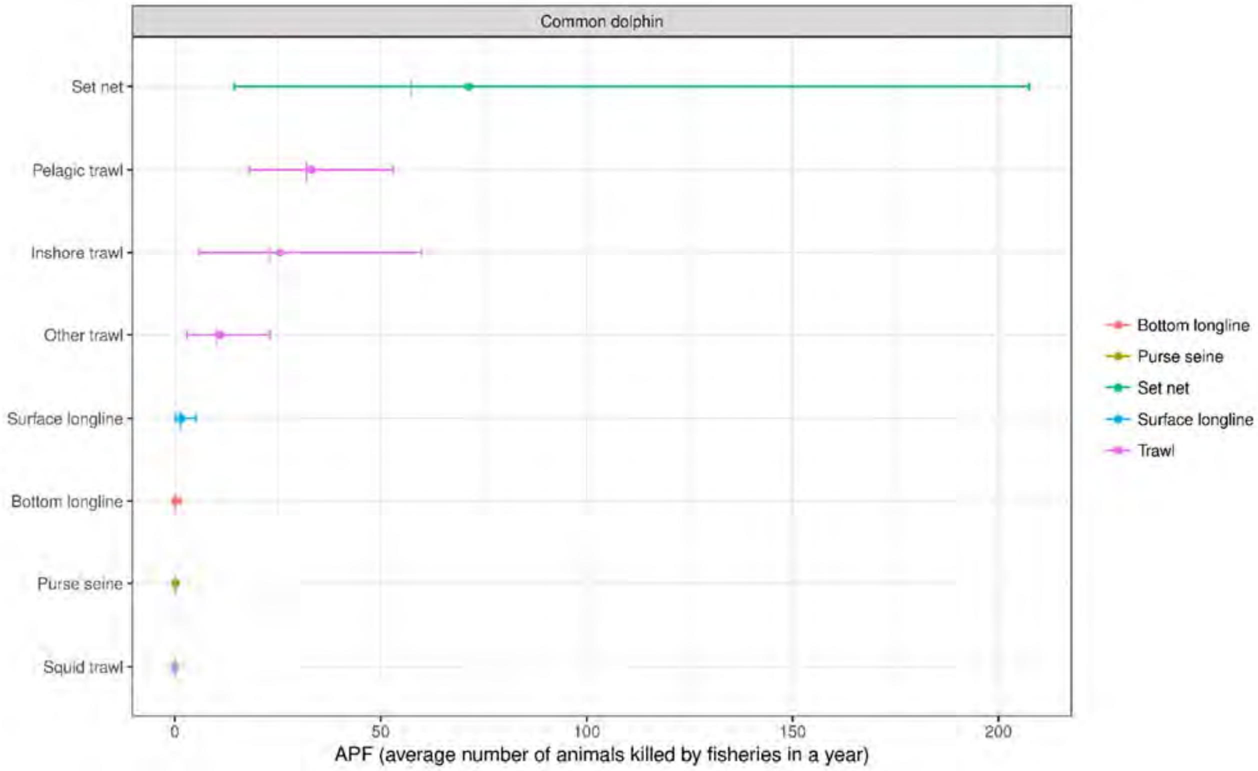


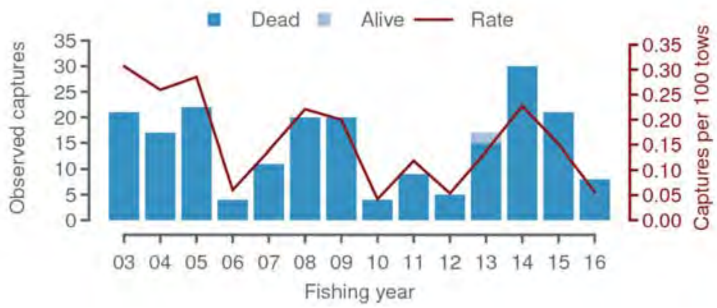
Figure 7.10: Annual fishery-related deaths of common dolphins in each fishery group, as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

7.4.5 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.

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]	46 estimated captures (95% c.i.: 32–67) in modelled trawl fisheries in 2014–15[4] 8 observed captures in trawl fisheries in 2015–16[4] 23 estimated captures (95% c.i.: 19–30) in the jack mackerel trawl fisheries in 2014–15[4] 19 observed captures in the jack mackerel trawl fisheries in 2014–15[4] 2 observed captures in the jack mackerel trawl fisheries in 2015–16[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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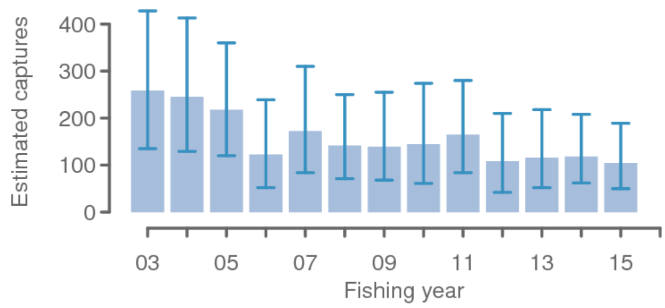
[1] Hammond et al. (2008).

[2] Baker et al. (2016).

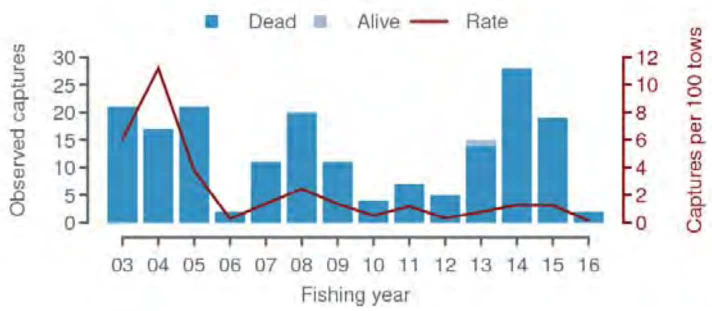
[3] Hammond et al. (2008).

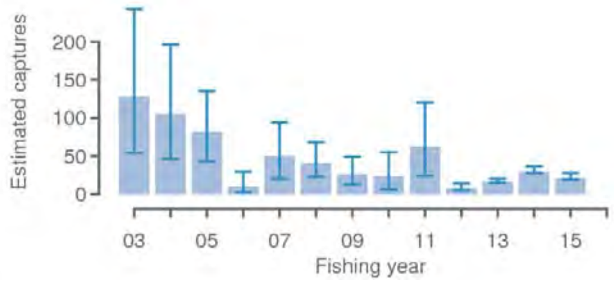
[4] For more information, see: http://data.dragonfly.co.nz/psc; note that at the time of publication, estimated captures for the 2015–16 year were not yet available.

[5] Abraham & Berkenbusch 2017.



Jack mackerel trawl fishery:





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8 NEW ZEALAND SEABIRDS

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 (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 or unobserved; species identity, especially of birds released alive; cryptic mortality rates; survival of birds released alive; 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. Consolidating qualitative and (semi) quantitative risk assessments is a key challenge.
Emerging issues	Assessing total fisheries impacts (i.e., including non-commercial and out-of-zone) and fisheries impacts in the context of other factors influencing seabird survival and reproduction, including other anthropogenic effects. Mortality caused by superstructure strikes. Potential new fishery monitoring techniques.
MPI research (current)	PRO2017-01A *Demographic parameters of black petrels;* PRO2017-01B *Demographic parameters of Southern Buller's*; PRO2017-05A *Population modelling of black petrels*; PRO2017-05B *Population modelling of Chatham Island Albatross*; PRO2017-06 *Characterisation of yellow-eyed penguin/fishery interactions*; PRO2017-15 *Innovative tag technology to examine foraging patterns of seabirds and association with fishing vessels*; PRO2017-19 *Capture rate of black petrels and flesh-footed shearwaters*; SEA2017-08 *A synthesis of the 'population' work in PRO2006-01*; SEA2017-10 *Black petrel electronic monitoring audit and analysis*; DAE2015-01 *Characterisation of seabird capture data*; PRO2015-01 *Improving estimates of cryptic mortality for seabird risk assessments*; PRO2014-06 *Update level-2 seabird risk assessment*.
NZ government research (current)	DOC Conservation Services Programme (CSP) projects: INT2017-02 *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,* INT2015-04 *Black petrel and flesh-footed shearwater foraging behaviour around fishing vessels,* POP2015-02 *Flesh-footed shearwater population*, POP2016-05 *Yellow-eyed penguin foraging and indirect effects*, POP2017-01 *Seabird population research: Chatham Island 2017-18,* POP2017-03 *Salvin's albatross: Bounties Islands population*, POP2017-04 *Seabird population research: Auckland Islands 2017-18,* POP2017-06 *Indirect effects on seabird in north-east North Island region,* MIT2016-01 *Protected species bycatch newsletter*, MIT2017-01 *Protected species liaison,* MIT2017-02 *Characterisation and development of offal management for small vessels*, MIT2017-03 *Characterisation and mitigation of protected species interactions in the inshore trawl fishery.*
Related chapters/issues	National Plan of Action (2013) to Reduce the Incidental Catch of Seabirds in New Zealand Fisheries (MPI 2013)

Note: This chapter has been updated for the 2017 AEBAR with recent population survey and capture estimation results.

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*, is the only species that is not protected. Black shag, *Phalacrocorax carbo*, and subantarctic skua, *Catharacta antarctica lonnbergi*, are partially protected, and sooty shearwater, *Puffinus griseus*, grey-faced petrel, *Pterodroma macroptera*, little shag, *Phalacrocorax melanoleucos brevirostris*, and pied shag, *Phalacrocorax varius*, may be hunted or killed subject to Minister's notification. Of the 85 seabird taxa that breed in New Zealand waters, 47 are considered threatened (by far the largest number in the world). 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 for Primary Industries 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 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).

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 2013) (http://www.fish.govt.nz/en-nz/Environmental/Seabirds/default.htm). The scopes of the 2004 and 2013 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 2013 NPOA covers all New Zealand fisheries and has a long-term objective that *'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.'* There are high-level subsidiary objectives related to practical aspects, biological risk, research and development, and international issues. Implementation is largely through MPI fisheries plans (see below). More detail is included in Section 8.4.3, *Managing fisheries interactions*.

Management of fishing-related mortality of seabirds is consistent 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 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, the objective most relevant for management of seabirds is Management Objective 2.5: *Manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened and protected species.*

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'*.

The Environment Objective is the same for all groups of fisheries in the draft National Fisheries Plan for Inshore Finfish and the draft National Fisheries Plan for Inshore Shellfish, to *'Minimise adverse effects of fishing on the aquatic environment, including on biological diversity'*. The draft National Fisheries Plan for Freshwater 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). 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 next page]

Common name	Scientific name	DOC category	IUCN category
Wandering albatross	*Diomedea exulans*	Non-Resident Native: Vagrant	Vulnerable
Antipodean albatross	*Diomedea antipodensis*	Threatened: Nationally Critical	#Vulnerable
Gibson's albatross	*Diomedea antipodensis*	Threatened: Nationally Critical	#Vulnerable
Southern royal albatross	*Diomedea epomophora*	At Risk: Naturally Uncommon	Vulnerable
Northern royal albatross	*Diomedea sanfordi*	At Risk: Naturally Uncommon	Endangered
Black-browed albatross	*Thalassarche melanophris*	Non-Resident Native: Coloniser	#Near
Campbell black-browed albatross	*Thalassarche impavida*	At Risk: Nationally Vulnerable	#Vulnerable
Southern Buller's albatross	*Thalassarche bulleri*	At Risk: Naturally Uncommon	#Near
Northern Buller's albatross	*Thalassarche bulleri platei.*	At Risk: Naturally Uncommon	#Near
White-capped albatross	*Thalassarche steadi**	At Risk: Naturally Uncommon	Near
Salvin's albatross	*Thalassarche salvini*	Threatened: Nationally Critical	Vulnerable
Chatham Island albatross	*Thalassarche eremita*	At Risk: Naturally Uncommon	Vulnerable
Eastern yellow-nosed albatross	*Thalassarche carteri*	Non-Resident Native: Coloniser	Endangered
Grey-headed albatross	*Thalassarche chrysostoma*	Threatened: Nationally Vulnerable	Endangered
Light mantled sooty albatross	*Phoebetria palpebrata*	At Risk: Declining	Near
Flesh-footed shearwater	*Puffinus carneipes*	Threatened: Nationally Vulnerable	Near
Wedge-tailed shearwater	*Puffinus pacificus*	At Risk: Relict	Least Concern
Buller's shearwater	*Puffinus bulleri*	At Risk: Naturally Uncommon	Vulnerable
Sooty shearwater	*Puffinus griseus*	At Risk: Declining	Near
Short-tailed shearwater	*Puffinus tenuirostris*	Non-Resident Native: Migrant	Least Concern
Fluttering shearwater	*Puffinus gavia*	At Risk: Relict	Least Concern
Hutton's shearwater	*Puffinus huttoni*	Threatened: Nationally Vulnerable	Endangered
Kermadec little shearwater	*Puffinus assimilis*	At Risk: Relict	#Least Concern
North Island little shearwater	*Puffinus assimilis*	At Risk: Recovering	#Least Concern
Subantarctic little shearwater	*Puffinus elegans*	At Risk: Naturally Uncommon	#Least Concern
Northern diving petrel	*Pelecanoides urinatrix*	At Risk: Relict	#Least Concern
Southern diving petrel	*Pelecanoides urinatrix*	At Risk: Relict	#Least Concern
Subantarctic diving petrel	*Pelecanoides urinatrix exsul*	Not Threatened	#Least Concern
South Georgian diving petrel	*Pelecanoides georgicus* †	Threatened: Nationally Critical	Least Concern
Grey petrel	*Procellaria cinerea*	At Risk: Naturally Uncommon	Near
Black petrel	*Procellaria parkinsoni*	Threatened: Nationally Vulnerable	Vulnerable
Westland petrel	*Procellaria westlandica*	At Risk: Naturally Uncommon	Vulnerable
White-chinned petrel	*Procellaria aequinoctialis*	Not Threatened	Vulnerable
Kerguelen petrel	*Lugensa brevirostris*	Non-Resident Native: Migrant	Least Concern
Southern Cape petrel	*Daption capense capense*	Non-Resident Native: Migrant	#Least Concern
Snares Cape petrel	*Daption capense australe*	At Risk: Naturally Uncommon	#Least Concern
Antarctic fulmar	*Fulmarus glacialoides*	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
Fairy prion	*Pachyptila turtur*	At Risk: Relict	Least Concern
Chatham fulmar prion	*Pachyptila crassirostris*	At Risk: Naturally Uncommon	#Least Concern
Lesser fulmar prion	*Pachyptila crassirostris*	At Risk: Naturally Uncommon	#Least Concern

Table 8.2 [Continued]:

Common name	Scientific name	DOC category	IUCN category
Thin-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
Broad-billed prion	*Pachyptila vittata*	At Risk: Relict	Least Concern
Blue petrel	*Halobaena caerulea*	Non-Resident Native: Migrant	Least Concern
Pycroft's petrel	*Pterodroma pycrofti*	At Risk: Declining	Vulnerable
Cook's petrel	*Pterodroma cookii*	At Risk: Relict	Vulnerable
Black-winged petrel	*Pterodroma nigripennis*	Not Threatened	Least Concern
Chatham petrel	*Pterodroma axillaris*	Threatened	Vulnerable
Mottled petrel	*Pterodroma inexpectata*	At Risk: Relict	Near Threatened
White-naped petrel	*Pterodroma cervicalis*	At Risk: Relict	Vulnerable
Kermadec petrel	*Pterodroma neglecta*	At Risk: Relict	Least Concern
Grey-faced petrel	*Pterodroma macroptera gouldi*	Not Threatened	Least Concern
Chatham Island taiko	*Pterodroma magentae*	Threatened: Nationally Critical	Critically
White-headed petrel	*Pterodroma lessonii*	Not Threatened	Least Concern
Soft-plumaged petrel	*Pterodroma mollis*	Non-Resident Native: Coloniser	Least Concern
Wilson's storm petrel	*Oceanites oceanicus*	Non-Resident Native: Migrant	Least Concern
Kermadec storm petrel	*Pelagodroma albiclunis*	Threatened: Nationally Critical	–
New Zealand storm petrel	*Pealeornis maoriana*	Threatened: Nationally	Critically
Grey-backed storm petrel	*Garrodia nereis*	At Risk: Relict	Least Concern
New Zealand white-faced storm	*Pelagodroma marina maoriana*	At Risk: Relict	#Least Concern
Black-bellied storm petrel	*Fregetta tropica*	Not Threatened	Least Concern
White-bellied storm petrel	*Fregetta grallaria grallaria*	Threatened: Nationally	Least Concern
Yellow-eyed penguin	*Megadyptes antipodes*	Threatened: Nationally	Endangered
Northern blue penguin**	*Eudyptula minor iredalei***	At Risk: Declining	#Least Concern
Southern blue penguin**	*Eudyptula minor minor***	At Risk: Declining	#Least Concern
Chatham Island blue penguin**	*Eudyptula minor chathamensis***	At Risk: Naturally Uncommon	#Least Concern
White-flippered blue penguin**	*Eudyptula minor albosignata***	Threatened: Nationally	#Least Concern
Eastern rockhopper penguin	*Eudyptes filholi*	Threatened: Nationally Critical	#Vulnerable
Fiordland crested penguin	*Eudyptes pachyrhynchus*	Threatened: Nationally	Vulnerable
Snares crested penguin	*Eudyptes robustus*	At Risk: Naturally Uncommon	Vulnerable
Erect-crested penguin	*Eudyptes sclateri*	At Risk: Declining	Endangered

* 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 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.

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 in (Table 8.1) which is further summarised here.

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 and blue penguin have been recorded nesting as yearlings and diving petrels and yellow-eyed penguins 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. Richard et al. (2011) list values for several demographic parameters that they used for a comprehensive seabird risk assessment. 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 penguins, and yellow-eyed penguins 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 20 years (Bartle 1991, Brothers 1991, Brothers et al. 1999, Croxall 2008) and the Food & Agriculture Organization 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. Croxall et al. (2012) reviewed the threats to 346 seabird taxa and concluded that: seabirds are more threatened than other comparable groups of birds; their status has deteriorated faster over recent decades; and fishing-related mortality is the most pervasive and immediate threat to many albatrosses and petrels. They listed the principal threats while at sea as being posed by commercial fisheries (through competition for food and mortality associated with fishing gear) and pollution, and those on land as being alien predators, habitat degradation and human disturbance. Direct exploitation, impacts of aquaculture, energy generation operations, and climate change were listed as threats for some taxa or areas where understanding was particularly poor.

Croxall et al. (2012) categorise responses to the issue of fishing-related mortality 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 spatiotemporal 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);
- working with multinational and international bodies (e.g., FAO and RFMOs) to develop and implement appropriate regulations for the use of best-practice techniques to reduce or eliminate seabird bycatch and;
- working with fishers (and national fishery organisations) to assist cost-effective implementation of these mitigation techniques.

Seabirds are ranked by the International Union for the Conservation of Nature (IUCN) as the world's most threatened bird grouping (Croxall et al. 2012). 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 the Agreement on the Conservation of Albatrosses and Petrels (ACAP) and DOC is the lead New Zealand agency. However, the key threat to seabirds at sea, especially albatrosses and petrels, is incidental capture and death in fisheries managed by MPI.

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., at 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 145 seabird taxa, second only to the USA (GA Taylor, Department of Conservation, personal communication). Roughly 95 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.

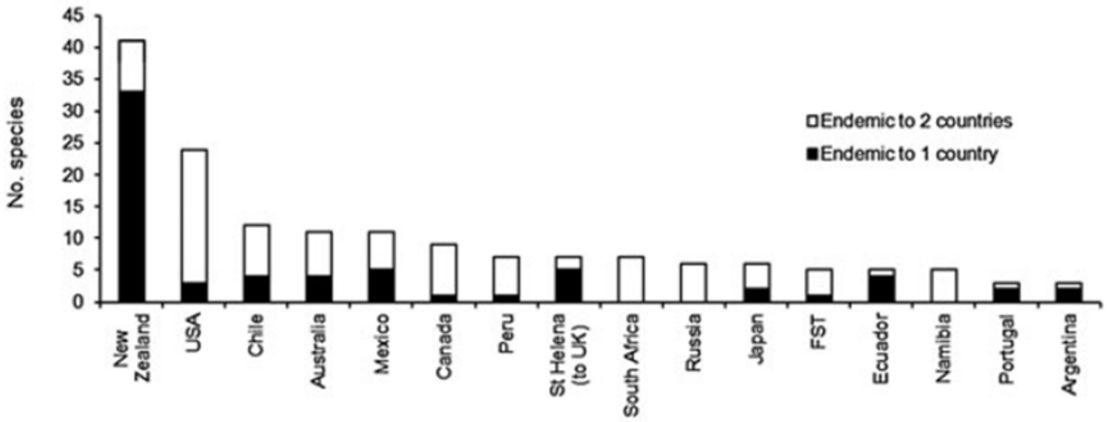


Figure 8.1: (from Croxall et al. 2012). Number of endemic breeding seabird taxa by country.

Some seabirds use New Zealand waters but do not breed here. Some visit here occasionally to feed (e.g., Indian Ocean yellow-nosed albatross and snowy wandering albatross), whereas others are frequent visitors (e.g., short-tailed shearwater and Wilson's storm petrel), sometimes for extended durations (e.g., juvenile giant petrels).

Taylor (2000) lists 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, Abraham et al. 2016) 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 and this was revised in 2013. This 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 somewhat by the time 2013 NPOA-seabirds was published but, 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), 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, foraging factors such as the season and areas they forage, their aggressiveness, 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 all forms of longlining. The nature and extent of interactions differs spatially, temporally, seasonally and diurnally between sectors, fisheries and between fleets and vessels within fisheries. In 2015–16 the taxa most frequently observed caught in New Zealand commercial fisheries in descending order were common diving petrel (288), white-chinned petrel (252), New Zealand white-capped albatross (156), southern Buller's albatross (115), sooty shearwater (63), Salvin's albatross (41), flesh-footed shearwater (21), Black petrel (20), Westland petrel (15), broad-billed prion (8), little penguin (8), grey petrel (5) and Antipodean albatross (5).

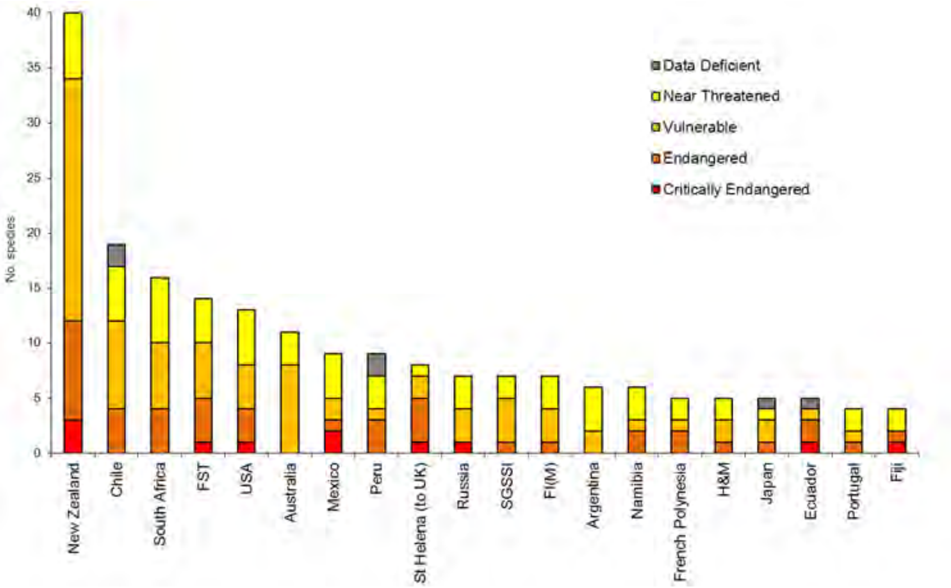


Figure 8.2: (from Croxall et al. 2012, supplementary material). The number of breeding and resident seabird species by country in each IUCN category (excluding Least Concern). FST, French Southern Territories; SGSSI, South Georgia and South Sandwich Islands; FI(M), Falkland Islands (Malvinas); H&M, Heard 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.

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

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, Abraham et al. 2016). 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.

In order 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. Unfortunately, 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 & Bell 2010, Francis et al. 2008, Dillingham & Fletcher 2011). For this reason, broad seabird risk assessments need to rely on expert knowledge (Level 1) or be semi-quantitative (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 were based on expert knowledge or were not comprehensive and 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 0.

8.4.1 SEABIRD DEMOGRAPHIC AND DISTRIBUTION STUDIES

This section summarises the key results of project PRO2006-01, *Demographic, distributional and trophic information on selected seabird species*, initiated by the Ministry of Fisheries (now MPI) to address some of the major information gaps on the demographics and distribution of seabird species commonly caught by commercial fishing in New Zealand waters. Other demographic studies have been conducted by the Department of Conservation or other parties and these are noted where possible.

8.4.1.1 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 with the hopes of establishing a second breeding site). In order 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 5245 (Fraser et al. 2011). This result compared closely with previous counts (which have ranged from 5194 to 5407 in late November and early December, Table 8.3) indicating a stable number of occupied nests on The Pyramid.

Chatham Island albatross 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 are given in Figure 8.3 (Fraser et al. 2010b).

To track the birds on a longer time-scale 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.

A full census of Chatham Island albatross was conducted in November 2016 yielding a similar population of 5296 nest sites, with the average from 1999–2016 being 5294 nest sites (range 5194–5407) (Bell et al. 2017). MPI has a current project to undertake population specific modelling of adult survival of the Chatham Island albatross (PRO2017-05B).

Table 8.3: (from Fraser et al. 2011). Counts of Chatham Island albatross nest sites for the years: 2007 (19–29 November); 2008 (22 November – 7 December); 2009 (9–12 December); and 2010 (24–30 December).

	2007	2008	2009	2010
Total nests counted	5 247	5 407	5 194	5 245

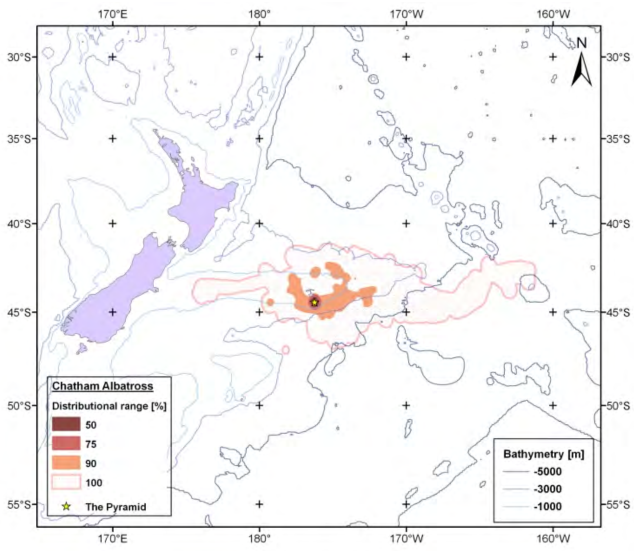


Figure 8.3: (from Fraser et al. 2010b). Distributional range of Chatham Island albatross during incubation and early chick rearing as derived from tracking studies in November/December 2007–09 (n=51 tracks).

8.4.1.2 NORTHERN BULLER'S ALBATROSS AND NORTHERN GIANT PETREL AT THE FORTY - FOURS, CHATHAM ISLANDS

The Forty-Fours, a small group of islands, are located about 35 km east of Chatham Island. They are home to the main breeding populations of northern royal albatross (*Diomedea sanfordi*) and northern Buller's albatross (*Thalassarche nov* sp.). A large colony of northern giant petrel (*Macronectes halli*) also breeds at the Forty-Fours. 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). Northern giant petrels nest mainly in the north-eastern part of the island along the cliff tops, interspersed with the northern royal albatross. Estimates of nests with chicks in them (both alive and dead) were: 430 in November 2007 (Scofield et al. 2008b); 349 in November 2008 (Fraser et al. 2009a); and 270 in December 2009 (Fraser et al. 2010a). Ten geolocators were placed on five incubating pairs of northern royal albatross in November 2007 (Scofield et al. 2008b).

In 2016, aerial photography was compared with ground counts to determine the population size of northern Buller's on the Forty-Fours and The Sisters. The aerial survey estimated 17 969 breeding pairs after correction with aerial close-ups and 16 138 breeding pairs after correction using ground counts (Baker et al. 2017).

A survey in December 2016 updated the population sizes for the northern Buller's albatross (17 682 nest sites) and northern giant petrel (1935 breeding pairs) (Bell et al. 2017).

8.4.1.3 NORTHERN ROYAL ALBATROSS

The main breeding populations of northern royal albatross are on the Forty-Fours and The Sisters, which are small island groups off the main Chatham Island. There is also a small colony at Taiaroa Head, South Island. The islands where northern royal albatross nest at the Chatham Islands are privately owned, and landing there is weather-dependent. In order to monitor populations effectively, counts are required immediately following egg laying (because this provides the most reliable estimates of the numbers of breeding pairs), and at fledging but prior to any chick departing each year (because this allows breeding success to be estimated each year). Aerial photography is the most cost-effective method of making these counts at these times and locations. Aerial counts of nesting northern royal albatross were made during each of the four breeding seasons 2006–07 to 2009–10.

Three trips to the Chatham Islands were planned each year during this study, with the primary objectives of each trip being to take aerial photographs for population counts on both the Forty-Fours and The Sisters. Trips were timed to coincide with key events in the breeding seasons and were planned for:

- Late November or early December (to count the number of northern royal albatrosses at the completion of egg laying);
- April (to count northern royal albatross chicks shortly after hatching); and
- September (to count northern royal albatross chicks just prior to fledging).

The November 2007 aerial survey was made just before the field team arrived on the Forty-Fours to study northern

Buller's albatross and northern giant petrels. A ground count of breeding northern royal albatross was made at about the same time of day as the aerial photography was completed. This one-off exercise showed that aerial and ground counts are broadly comparable and there is probably little bias caused by birds being obscured to aerial counting or the counting of non-breeding birds. Aerial counts suggested that the estimated total number of breeding pairs ranged from 5388 to 5744 (Table 8.4). These estimates do not differ markedly from an estimate made in the 1970s (Robertson 1998, cited in Scofield 2011).

At the small population that self-established on the mainland of New Zealand at Taiaroa Head, banding as well as monitoring of individuals has been carried out since 1938. Richard & Abraham (2013b) estimated the overall annual adult survival rate at 0.95 (95% c.i.: 0.941–0.959). Estimates of other demographic rates were also obtained during the estimation process. The mean age at first return of juveniles to the colony was estimated at 4.81 years (95% c.i.: 4.63–5.06), and the mean age at first breeding as 8.85 years (95% c.i.: 8.53–9.29).

In December 2016, a survey was conducted on the Forty-Fours that estimated 1400 incubating eggs of northern royal albatross, significantly lower than seen between 2006–07 and 2009–10 (Bell et al. 2017). The Sisters were not able to be surveyed due to poor weather conditions in 2016–17.

Table 8.4: (from Scofield 2011). Aerial counts of northern royal albatross eggs and chicks at their key Chatham Islands nesting sites, 2006–07 to 2009–10.

	2006–07		2007–08		2008–09		2009–10	
	Eggs	Chicks	Eggs	Chicks	Eggs	Chicks	Eggs	Chicks
Forty-Fours	1 879	1 018	2 212	1 093	2 055	1 036	2 692	1 083
Big Sister	2 128	871	2 018	288	2 081	496	1 893	665
Middle Sister	1 381	670	1 371	435	1 316	483	1 159	569
Total	5 388	2 559	5 601	1 816	5 452	2 015	5 744	2 317

8.4.1.4 SALVIN'S ALBATROSS ON BOUNTY ISLANDS

Salvin's albatross (*Thalassarche salvini*) is endemic to New Zealand, breeding only on the Bounty Islands and the Western Chain of The Snares. 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) as 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 to the corrected estimate for 2010 of 31 786 (95% c.i: 31 430–32 143).

DOC Conservation Services Programme have been reviewing the methodology for undertaking a survey of the Salvin's albatross on the Bounty islands (Debski & Hjörvarsdóttir 2017).

8.4.1.5 SALVIN'S ALBATROSS ON SNARES WESTERN CHAIN

In 2008, a three-year study of Salvin's albatrosses was initiated at the Snares Western Chain. 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 and Rima Islets during October 2008 (Charteris et al. 2009) and September–October 2009, respectively (Carroll et al. 2010) (Table 8.5). Only Toru Island was sampled in 2010.

Table 8.5: (from Sagar et al. 2011). Numbers of Salvin's albatross pairs breeding on Toru and Rima Isles, Western Chain, The Snares, 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

In order 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).

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 presented in Figure 8.4. 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 in eastern Bass Strait and along the eastern coast of Tasmania (Figure 8.4a). 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 occurred primarily east of the Chatham Islands, off Puysegur and on the Stewart-Snares Shelf (Figure 8.4b). Eggs are laid starting in August and all of the birds occurred within Australasian waters throughout August to October, primarily on the Challenger Plateau, off Puysegur, the Stewart-Snares Shelf, and Campbell Plateau (Figure 8.4c). During this period these birds from the Snares 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, the Stewart-Snares Shelf, and Campbell Plateau (Figure 8.4d). 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. 2015). 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).

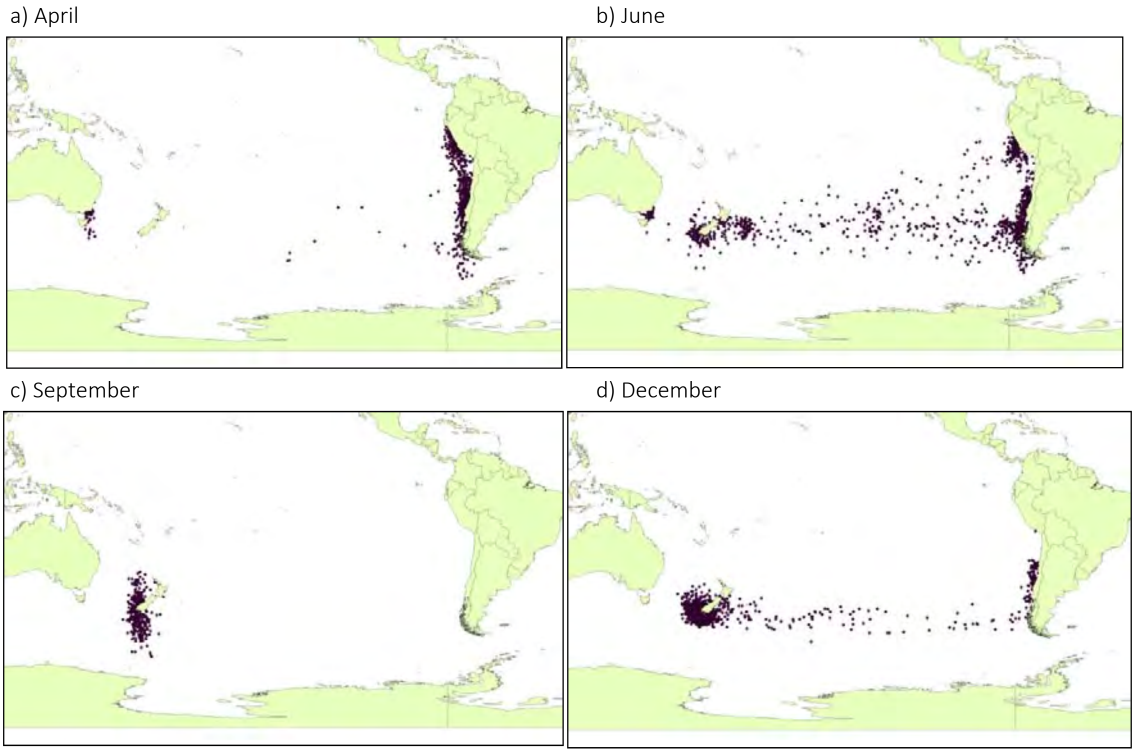


Figure 8.4: (from Sagar et al. 2011). Distribution of Salvin's albatrosses *Thalassarche salvini* from the Snares Western Chain tagged with geolocators at four times of the year: a) April, after the completion of their breeding season, b) June, showing their return tracks from South American waters to 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, whilst the birds crossing the Pacific and in South American waters are presumed to be failed breeders.

8.4.1.6 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: 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 8.6). 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).

2011–14: Surveys suggested 99 776 breeding pairs in 2011, 118 098 in 2012, 95 278 in 2013 and 101 798 in 2014. 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) to 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

8.6). 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).

Further aerial surveys were conducted in 2016 (Baker & Jensz 2016) and 2017. The resulting aerial photographs are being analysed under MPI project SEA2016-29.

A marked population of breeding adult white-capped albatross has been established over 2015–17 in order to estimate their demographic parameters in the long term, including adult survival (Parker et al. 2016).

Table 8.6: (after Baker et al. 2013, 2014b, 2015a). Aerial-photographic counts of breeding pairs of white-capped albatrosses on three islands in the Auckland Islands group in December 2006–14.

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
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

8.4.1.7 WHITE-CHINNED PETREL ON ANTIPODES ISLANDS

In 2007, a five-year study of white-chinned petrels (*Procellaria aequinoctialis*) was initiated on Antipodes Island. Four seasons of fieldwork have been completed (Sommer et al. 2008, 2009, 2010). The objectives of the white-chinned petrel field work were to:

- estimate the population trend from mark-recapture in the three study areas;
- determine foraging locations and activity; and
- estimate burrow occupancy in a range of habitats in order to increase the accuracy of a total island population estimate.

Three study areas were established and all white-chinned petrel burrows in each were checked at least three times during each field trip to identify both birds. Identifying white-chinned petrel burrows can involve a degree of subjectivity because white-headed petrels, *Pterodroma lessoni*, also nest on Antipodes Island. Although many white-chinned petrel burrows have very large entrances, and many white-headed petrel burrows have much smaller entrances with steep tunnels, white-chinned petrel have been found in burrows with entrances that have characteristics somewhere between the two. Estimated occupancy rates were similar in the years studied (Table 8.7). Overall, the number of burrows fluctuates between years as new burrows are dug and the number of burrows with unidentified eggshell varies (Sommer et al. 2010).

Table 8.7: (from Sommer et al. 2010). White-chinned petrel (WCP) study burrow occupancy between years.

Year	Timing	Total 'WCP' burrows counted	'WCP' burrows with breeding WCP	% with breeding WCP
2008	mid Jan to end Feb	280	71	25.4
2009	late Jan to end Feb	285	77	27.0
2010	mid Dec to early Jan	295	81	27.5

To determine the foraging area of breeding white-chinned petrels, 34 dataloggers (30 British Antarctic Survey, 4 Lotek) were deployed on breeding white-chinned petrels in 2008 (Sommer et al. 2008). Seventeen and 13 of these birds were recaptured during the 2009 and 2010 field trips and their dataloggers were removed (Sommer et al. 2009, 2010). Data from the 17 geolocators recovered during 2009 have been processed and enable initial conclusions to be made of the foraging movements of white-chinned petrels from the Antipodes. In summary, these are:

- During the breeding season, the birds foraged within the EEZ, mostly north of Antipodes Island and to the east of the mainland (Figure a).
- There was movement of birds across the Pacific to the coasts of Chile and Peru during February, presumably by failed breeders (Figure 8.5b).
- In the latter part of the breeding season (April and May) the birds tended to forage south of Antipodes Island.
- In May, after breeding, all birds migrated across the Pacific to forage off the west coast of South America, remaining there until August (Figure 8.5c).
- In September, the birds returned across the Pacific to Antipodes Island from the coast of Peru for the start of the new breeding season.

Occupancy was also estimated across a range of habitats throughout the island using transects. These transects varied in length and were measured by saving tracks on a handheld GPS. All white-chinned petrel burrows within 1 m either side of the transect (i.e., a 2 m-wide strip in total) were recorded (Table 8.8) and occupancy determined using a stick or burrow scope. Habitat type and slope were also recorded for each burrow (Sommer et al. 2008, 2009, 2010).

Between December 2009 and January 2010, breeding white-chinned petrels were estimated to have an average density across all sampled habitats of 45 occupied burrows.ha^{-1}. The total area of Antipodes Island is 2025 ha (Bell 2002) and, assuming all of this area is similarly suitable to the sampled areas, a preliminary estimate of the total population is 91 125 breeding pairs (Sommer et al. 2010), compared with 100 000 pairs estimated by Taylor (2000). Habitat information (slope, aspect, vegetation) has been recorded for each transect and a quantitative survey of the extent of different habitat types over the entire area was completed during the 2011 field season to allow a more robust population estimate to be calculated, based on burrow densities in different habitat types.

Rexer-Huber et al. (2015), based on a line transect survey conducted in January 2015, estimated the population of white-chinned petrels on Disappointment Island at 153 100 (115 900–202 200) breeding pairs.

Table 8.8: (from Sommer et al. 2010). Results of white-chinned petrel occupancy transects in various habitats spread throughout Antipodes Island.

No. transects	Total burrows	No. containing white-chinned petrel breeding (non-breeding)	No. containing white-headed petrel	No. empty	No. not used for occupancy estimate	% burrows with breeding white-chinned petrel
20	247	59 (10)	21	144	13	25.2

a) December

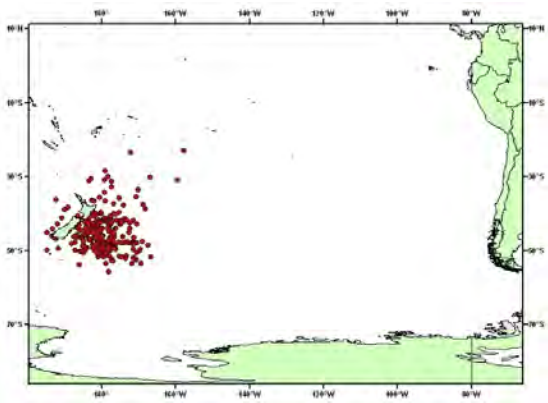
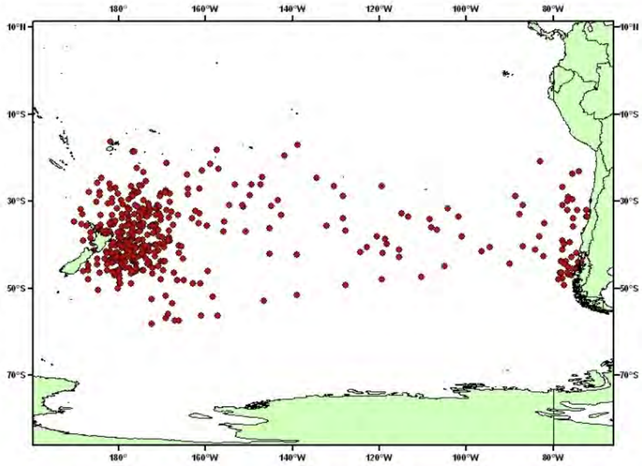


Figure 8.5: (from Sommer et al. 2010). Foraging locations of white-chinned petrels from the Antipodes, in a) December, b) February and in c) June–August, after the end of the breeding season. [Continued on next page]

b) February



c) June–August

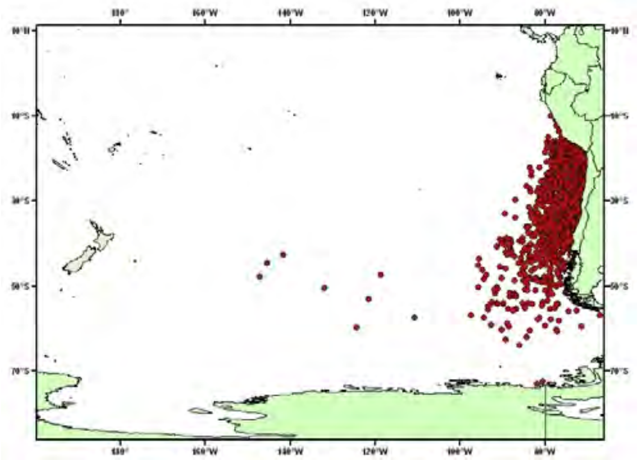


Figure 8.5 [Continued]: (from Sommer et al. 2010). Foraging locations of white-chinned petrels from the Antipodes, in a) December, b) February and in c) June–August, after the end of the breeding season.

8.4.1.8 GREY PETREL ON ANTIPODES ISLANDS

A two-year study of grey petrels (*Procellaria cinerea*) on Antipodes Island commenced during 2009 and was completed during the period 19 March–30 April 2010. The objectives of the grey petrel field work were to:

- estimate the population trend from mark-recapture analysis in the study areas;
- determine foraging location and activity; and
- estimate the total island population by examining burrow occupancy in a range of habitats.

In 2009, a total of 69 burrows in Alert Bay, the Crater and Crater Ridge containing grey petrels were marked as study burrows (Sommer et al. 2009). In addition, 64 grey petrel burrows within the white-chinned petrel study areas were used as study burrows (Sommer et al. 2010).

To establish the foraging distribution of grey petrels, 27 geolocation dataloggers were deployed on breeding grey petrels in 2009 (Sommer et al. 2009). Eighteen of the 27 geolocators deployed were subsequently retrieved, although one datalogger had dislodged from the attachment to the petrel. Data from the geolocators are being processed by the British Antarctic Survey (Sommer et al. 2010).

Occupancy transects were carried out after peak egg laying in the study burrows. Because of the short daylight hours at this time of year transects were limited to the northern half of the island. Transects were conducted in all habitat types on the coastal and inland slopes. A few transects were also done on the flatter ground more usually associated with white-chinned petrels. Transects were mapped and measured by recording the position of the start and end of each transect as well as each burrow with a hand-held GPS.

Sommer et al. (2010) estimated a breeding population of 48 960 pairs (96 pairs.ha^{-1} over 510 ha of suitable habitat). Although two seasons of field work on grey petrels is insufficient to allow an assessment of population trend over this period, a comparison of population trend is possible with reference to the earlier study of Bell (2002) who reported a mean of 104 occupied grey petrel burrows ha^{-1} from a survey completed during April–June 2001. Assuming the same 510 ha of suitable habitat on Antipodes Island, Bell estimated a breeding population of 53 040 pairs, similar to Sommer et al.'s (2010) estimate.

Parker et al. (2015) surveyed Campbell Island to assess whether the grey petrels have increased in number following the eradication of rats in 2001. They found that the population of grey petrels remains small, at 90 pairs from the four colonies located.

8.4.1.9 FLESH-FOOTED SHEARWATER

Flesh-footed shearwaters, *Puffinus carneipes*, 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, cited in Baker et al. in prep) 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 prep.). Field work was focused on visiting all of the 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, in prep.) 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 search transects were established by following a compass bearing downhill from the ridgeline. When potential burrows were located, their location of that colony 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.

All colony areas, with the exception of those on Karewa, 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 encountered.

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 8.9). Baker et al. (in prep.) estimate 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. (in prep.) state that their estimates generally accord with the indicative population estimates developed by Graeme Taylor (cited in Baker et al. in prep.) with the exception of that for Coppermine and Ohinau Islands. Baker et al.'s (in prep.) estimate of 1425 occupied burrows (1059–1791) for Coppermine is much lower than Taylor's indicative estimate of 10 000 (presumably breeding pairs). In contrast, Baker et al.'s (in prep.) estimate of 2071 occupied burrows (943–3200) for Ohinau greatly exceeds Taylor's indicative estimate.

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/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.

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).

Table 8.9: (from Baker et al. in prep.). Estimated number of potential and occupied burrows for flesh-footed shearwater for eight New Zealand islands surveyed 2007–08 to 2010–11. Note that some colonies on Lady Alice and Coppermine 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 2007–10 (Baker et al. 2008b, 2009a, 2010a, in prep.) 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/Mauimua (Waugh et al. 2014).

Tracking of flesh-footed shearwaters 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. 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).

8.4.1.10 WESTLAND PETREL

The Westland petrel, *Procellaria westlandica*, is endemic to New Zealand and nests in burrows in dense rainforest near Punakaiki, Westland. This species is poorly studied, probably largely because they 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 mapped burrows 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 by the density to arrive at a population estimate for each colony.

Although Westland petrels breed throughout a 16 square kilometre 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). Baker et al. (2007b, 2008a, 2011a) estimated the number of potential burrows in all Westland petrel colonies to total 6846 (95% c.i.: 6389–7302) during the period 2007 to 2011. 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.

8.4.2 QUANTIFYING FISHERIES INTERACTIONS

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 MPI 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., Abraham et al. 2010b, Abraham & Thompson 2009a, 2010, 2011a, 2011b, Ayers et al. 2004, Baird 1994, 1995, 1996, 1997, 1999, 2000, 2001a, 2001b, 2003, 2004a–c, 2005, Baird et al. 1998, 1999, Baird & Griggs 2004, Thompson & Abraham 2009). 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 or photographs) and description of biological characters (sex, age, condition, etc., available only from carcasses).

In some fisheries, observer data are temporally and spatially well stratified, whilst 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 in Abraham et al. (2016). In this context, captures include all seabirds recovered on a fishing vessel except birds that simply 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). Observer coverage has been highly heterogeneous 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 8.6).

Fisher-reported captures (on NFPSCR forms available since 1 October 2008) have not been used to estimate total captures because the reported capture rates are much lower than those reported by independent observers (Abraham & Thompson 2011b) and the species identification is 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 (*Puffinus griseus*); white-chinned petrel (*Procellaria aequinoctialis*); black petrel (*Procellaria parkinsoni*); grey petrel (*Procellaria cinerea*); white-capped albatross (*Thalassarche steadi*); Salvin's albatross (*Thalassarche salvini*); southern Buller's albatross (*Thalassarche bulleri*); 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 or Antipodean wandering albatrosses or Campbell Island albatrosses. The 'other birds' category includes many taxa but grey, black, great-winged, and Cape petrels (both subspecies but mostly Southern Cape petrels, *Daption capense capense*), flesh-footed shearwater, and spotted shag 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 2015–16 year are shown in Table 8.10 to Table 8.18.

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 8.7). 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 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 8.19). 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 8.8), again largely reflecting the distribution of the major commercial fisheries (for southern bluefin and other tunas). The charter fleet targeting tuna has historically had much higher observer coverage than the domestic fleet. Southern Buller's and white-capped 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 8.20). 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 have been the most frequently observed captured of the albatross species, and white-chinned petrel, flesh-footed shearwater, grey petrel, sooty shearwater, and black petrels have been the other species most frequently observed (Table 8.21). Only about 14% of observed captures were albatrosses.

Model-based estimates of captures can be combined across trawl and longline fisheries (Figure 8.10). 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 2015–16) 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 longliners. 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 albatrosses) combined, with 44% and 43% taken in surface- and bottom-longline fisheries, respectively. Bottom longline and trawl take similar proportions of the estimated total captures of white-chinned petrels (29% and 66%, respectively).

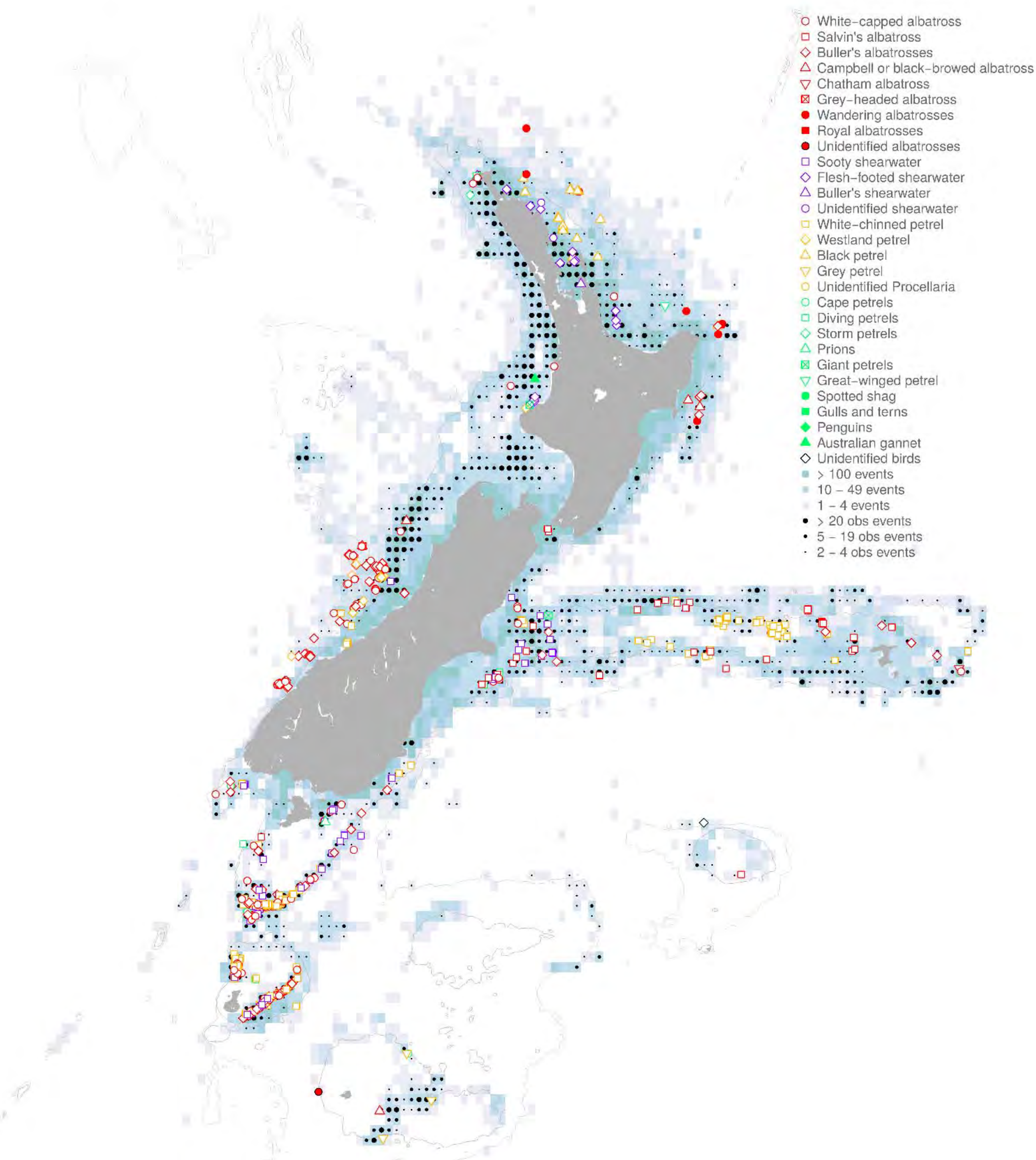


Figure 8.6: All observed seabird captures in trawl, surface-longline, bottom-longline, set-net and purse seine fishing within New Zealand region, between October and September 2015. The colour within each 0.2 degree cell indicates the number of fishing events (tows and sets, darker colours indicate more fishing) and the black dots indicate the number of observed 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. http://data.dragonfly.co.nz/psc. Data version v2017001.

Table 8.10: Summary of observed and model-estimated total captures of all seabirds combined by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017001.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	261	3.82	3 201	2 687–3 870	100.0
2003–04	120 850	6 549	5.4	251	3.83	2 743	2 316–3 270	100.0
2004–05	120 461	7 713	6.4	434	5.63	3 437	2 949–4 003	100.0
2005–06	109 951	6 619	6.0	345	5.21	3 125	2 619–3 813	100.0
2006–07	103 317	7 934	7.7	208	2.62	2 219	1 827–2 684	100.0
2007–08	89 536	9 050	10.1	232	2.56	1 927	1 621–2 320	100.0
2008–09	87 550	10 011	11.4	460	4.59	2 266	1 947–2 664	100.0
2009–10	92 889	9 525	10.3	264	2.77	2 058	1 739–2 463	100.0
2010–11	86 091	7 583	8.8	392	5.17	2 311	1 970–2 711	100.0
2011–12	84 424	9 336	11.1	248	2.66	1 797	1 516–2 159	100.0
2012–13	83 832	12 401	14.8	712	5.74	2 107	1 845–2 447	100.0
2013–14	85 114	13 203	15.5	488	3.70	1 883	1 619–2 212	100.0
2014–15	78 757	13 869	17.6	620	4.47	1 987	1 733–2 295	100.0
2015–16	78 040	14 181	18.2	462	3.26	1 695	1 462–1 999	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	115	0.052	2 542	1 828–3 843	100.0
2003–04	7 386 339	1 607 304	21.8	71	0.044	1 876	1 337–2 967	100.0
2004–05	3 679 965	783 812	21.3	41	0.052	927	624–1 598	100.0
2005–06	3 691 809	705 945	19.1	37	0.052	932	657–1 414	100.0
2006–07	3 740 012	1 040 948	27.8	187	0.18	860	660–1 197	100.0
2007–08	2 246 689	421 900	18.8	37	0.088	564	399–831	100.0
2008–09	3 114 733	937 496	30.1	57	0.061	703	506–1 045	100.0
2009–10	2 996 544	673 333	22.5	145	0.215	857	656–1 166	100.0
2010–11	3 186 899	674 572	21.2	47	0.07	833	588–1 238	100.0
2011–12	3 100 277	728 190	23.5	66	0.091	856	623–1 214	100.0
2012–13	2 876 932	560 333	19.5	27	0.048	807	582–1 158	100.0
2013–14	2 549 764	782 541	30.7	36	0.046	682	486–994	100.0
2014–15	2 412 336	725 370	30.1	38	0.052	589	414–869	100.0
2015–16	2 359 891	322 960	13.7	131	0.406	840	632–1 158	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	257	0.024	2 897	2 102–4 259	100.0
2003–04	43 243 060	5 038 339	11.7	57	0.011	2 359	1 637–3 700	100.0
2004–05	41 855 058	2 883 725	6.9	30	0.01	2 908	1 910–4 803	100.0
2005–06	37 152 933	3 802 951	10.2	41	0.011	2 297	1 502–3 891	100.0
2006–07	38 129 760	2 315 772	6.1	59	0.025	2 967	1 882–5 138	100.0
2007–08	41 512 898	3 593 511	8.7	40	0.011	2 528	1 652–4 285	100.0
2008–09	37 437 488	4 052 896	10.8	39	0.01	2 309	1 523–3 842	100.0
2009–10	40 441 911	2 683 143	6.6	58	0.022	2 436	1 614–4 065	100.0
2010–11	40 902 266	1 740 035	4.3	20	0.011	2 608	1 733–4 329	100.0
2011–12	37 894 023	2 104 731	5.6	10	0.005	2 269	1 488–3 876	100.0
2012–13	32 508 639	801 318	2.5	6	0.007	1 990	1 318–3 348	100.0
2013–14	40 872 925	3 094 416	7.6	105	0.034	2 248	1 546–3 535	100.0
2014–15	39 339 271	742 291	1.9	27	0.036	2 004	1 325–3 258	100.0
2015–16	43 490 620	3 148 850	7.2	104	0.033	1 983	1 343–3 197	100.0

Table 8.11: Summary of observed and model-estimated total captures of white-capped albatross by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	83	1.21	700	538–907	100.0
2003–04	120 850	6 549	5.4	138	2.11	813	634–1 031	100.0
2004–05	120 461	7 713	6.4	213	2.76	1 010	806–1 254	100.0
2005–06	109 951	6 619	6	63	0.95	553	417–723	100.0
2006–07	103 317	7 934	7.7	54	0.68	439	324–575	100.0
2007–08	89 536	9 050	10.1	44	0.49	329	243–441	100.0
2008–09	87 550	10 011	11.4	90	0.9	425	333–539	100.0
2009–10	92 889	9 525	10.3	42	0.44	341	247–458	100.0
2010–11	86 091	7 583	8.8	44	0.58	352	257–468	100.0
2011–12	84 424	9 336	11.1	70	0.75	378	287–495	100.0
2012–13	83 832	12 401	14.8	134	1.08	383	298–491	100.0
2013–14	85 113	13 203	15.5	79	0.6	319	239–424	100.0
2014–15	78 754	13 869	17.6	75	0.54	281	212–371	100.0
2015–16	78 040	14 181	18.2	108	0.76	330	256–426	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	2	0.001	86	38–164	100.0
2003–04	7 386 339	1 607 304	21.8	17	0.011	139	73–244	100.0
2004–05	3 679 965	783 812	21.3	3	0.004	33	13–70	100.0
2005–06	3 691 809	705 945	19.1	2	0.003	31	11–66	100.0
2006–07	3 740 012	1 040 948	27.8	28	0.027	47	34–68	100.0
2007–08	2 246 689	421 900	18.8	3	0.007	36	13–78	100.0
2008–09	3 114 733	937 496	30.1	3	0.003	41	16–87	100.0
2009–10	2 996 544	673 333	22.5	32	0.048	81	51–135	100.0
2010–11	3 186 899	674 572	21.2	3	0.004	50	20–103	100.0
2011–12	3 100 277	728 190	23.5	8	0.011	140	67–264	100.0
2012–13	2 876 932	560 333	19.5	12	0.021	129	64–235	100.0
2013–14	2 549 764	782 541	30.7	7	0.009	104	48–199	100.0
2014–15	2 412 336	725 370	30.1	7	0.01	100	47–191	100.0
2015–16	2 359 891	322 960	13.7	39	0.121	143	84–244	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	0	0	24	5–56	100.0
2003–04	43 243 060	5 038 339	11.7	1	0	21	5–51	100.0
2004–05	41 855 058	2 883 725	6.9	0	0	27	6–63	100.0
2005–06	37 152 933	3 802 951	10.2	1	0	20	5–44	100.0
2006–07	38 129 760	2 315 772	6.1	0	0	27	6–65	100.0
2007–08	41 512 898	3 593 511	8.7	0	0	26	6–61	100.0
2008–09	37 437 488	4 052 896	10.8	0	0	26	6–61	100.0
2009–10	40 441 911	2 683 143	6.6	0	0	23	5–55	100.0
2010–11	40 902 266	1 740 035	4.3	0	0	33	8–78	100.0
2011–12	37 894 023	2 104 731	5.6	2	0.001	29	8–66	100.0
2012–13	32 508 639	801 318	2.5	0	0	25	5–59	100.0
2013–14	40 872 925	3 094 416	7.6	0	0	27	6–64	100.0
2014–15	39 339 271	742 291	1.9	1	0.001	24	6–53	100.0
2015–16	43 490 620	3 148 850	7.2	0	0	24	5–56	100.0

Table 8.12: Summary of observed and model-estimated total captures of Salvin's albatross by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	22	0.32	480	317–700	100.0
2003–04	120 850	6 549	5.4	11	0.17	385	240–589	100.0
2004–05	120 461	7 713	6.4	33	0.43	621	414–913	100.0
2005–06	109 951	6 619	6	7	0.11	363	230–547	100.0
2006–07	103 317	7 934	7.7	11	0.14	323	207–488	100.0
2007–08	89 536	9 050	10.1	7	0.08	229	139–359	100.0
2008–09	87 550	10 011	11.4	36	0.36	306	204–453	100.0
2009–10	92 889	9 525	10.3	43	0.45	349	239–494	100.0
2010–11	86 091	7 583	8.8	20	0.26	318	210–464	100.0
2011–12	84 424	9 336	11.1	26	0.28	306	206–446	100.0
2012–13	83 832	12 401	14.8	52	0.42	348	242–491	100.0
2013–14	85 113	13 203	15.5	47	0.36	369	258–527	100.0
2014–15	78 754	13 869	17.6	46	0.33	375	266–529	100.0
2015–16	78 040	14 181	18.2	34	0.24	291	202–410	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	0	0	48	20–86	100.0
2003–04	7 386 339	1 607 304	21.8	0	0	37	15–69	100.0
2004–05	3 679 965	783 812	21.3	0	0	15	5–31	100.0
2005–06	3 691 809	705 945	19.1	0	0	19	7–37	100.0
2006–07	3 740 012	1 040 948	27.8	1	0.001	17	7–33	100.0
2007–08	2 246 689	421 900	18.8	2	0.005	12	5–22	100.0
2008–09	3 114 733	937 496	30.1	3	0.003	16	7–29	100.0
2009–10	2 996 544	673 333	22.5	1	0.001	17	6–32	100.0
2010–11	3 186 899	674 572	21.2	0	0	11	3–22	100.0
2011–12	3 100 277	728 190	23.5	1	0.001	8	3–17	100.0
2012–13	2 876 932	560 333	19.5	0	0	9	2–18	100.0
2013–14	2 549 764	782 541	30.7	0	0	6	1–14	100.0
2014–15	2 412 336	725 370	30.1	0	0	5	1–11	100.0
2015–16	2 359 891	322 960	13.7	0	0	6	1–14	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	14	0.001	88	31–244	100.0
2003–04	43 243 060	5 038 339	11.7	9	0.002	97	32–268	100.0
2004–05	41 855 058	2 883 725	6.9	0	0	109	26–320	100.0
2005–06	37 152 933	3 802 951	10.2	1	0	90	19–305	100.0
2006–07	38 129 760	2 315 772	6.1	22	0.01	132	44–374	100.0
2007–08	41 512 898	3 593 511	8.7	0	0	111	24–346	100.0
2008–09	37 437 488	4 052 896	10.8	1	0	107	28–293	100.0
2009–10	40 441 911	2 683 143	6.6	0	0	122	29–365	100.0
2010–11	40 902 266	1 740 035	4.3	2	0.001	128	34–382	100.0
2011–12	37 894 023	2 104 731	5.6	0	0	141	33–439	100.0
2012–13	32 508 639	801 318	2.5	0	0	141	34–418	100.0
2013–14	40 872 925	3 094 416	7.6	6	0.002	167	52–460	100.0
2014–15	39 339 271	742 291	1.9	0	0	118	29–344	100.0
2015–16	43 490 620	3 148 850	7.2	6	0.002	139	44–364	100.0

Table 8.13: Summary of observed and model-estimated total captures of Buller's albatross by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version 2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	6	0.09	118	59–203	100.0
2003–04	120 850	6 549	5.4	9	0.14	131	70–216	100.0
2004–05	120 461	7 713	6.4	24	0.31	187	120–277	100.0
2005–06	109 951	6 619	6	10	0.15	127	71–207	100.0
2006–07	103 317	7 934	7.7	6	0.08	96	51–166	100.0
2007–08	89 536	9 050	10.1	18	0.2	120	75–184	100.0
2008–09	87 550	10 011	11.4	18	0.18	96	58–150	100.0
2009–10	92 889	9 525	10.3	11	0.12	90	49–153	100.0
2010–11	86 091	7 583	8.8	22	0.29	107	68–159	100.0
2011–12	84 424	9 336	11.1	36	0.39	140	97–200	100.0
2012–13	83 832	12 401	14.8	59	0.48	123	91–175	100.0
2013–14	85 113	13 203	15.5	37	0.28	106	73–157	100.0
2014–15	78 754	13 869	17.6	35	0.25	105	73–151	100.0
2015–16	78 040	14 181	18.2	56	0.39	135	99–185	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	41	0.019	452	257–762	100.0
2003–04	7 386 339	1 607 304	21.8	39	0.024	391	225–671	100.0
2004–05	3 679 965	783 812	21.3	22	0.028	139	76–249	100.0
2005–06	3 691 809	705 945	19.1	15	0.021	156	81–287	100.0
2006–07	3 740 012	1 040 948	27.8	47	0.045	145	93–231	100.0
2007–08	2 246 689	421 900	18.8	20	0.047	103	58–180	100.0
2008–09	3 114 733	937 496	30.1	30	0.032	132	75–227	100.0
2009–10	2 996 544	673 333	22.5	67	0.1	180	123–272	100.0
2010–11	3 186 899	674 572	21.2	28	0.042	141	84–232	100.0
2011–12	3 100 277	728 190	23.5	33	0.045	189	110–320	100.0
2012–13	2 876 932	560 333	19.5	10	0.018	139	74–245	100.0
2013–14	2 549 764	782 541	30.7	23	0.029	135	77–234	100.0
2014–15	2 412 336	725 370	30.1	21	0.029	117	65–207	100.0
2015–16	2 359 891	322 960	13.7	57	0.176	186	120–303	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	1	0	32	11–69	100.0
2003–04	43 243 060	5 038 339	11.7	0	0	31	11–63	100.0
2004–05	41 855 058	2 883 725	6.9	0	0	53	20–107	100.0
2005–06	37 152 933	3 802 951	10.2	0	0	46	16–98	100.0
2006–07	38 129 760	2 315 772	6.1	0	0	53	18–112	100.0
2007–08	41 512 898	3 593 511	8.7	6	0.002	75	33–145	100.0
2008–09	37 437 488	4 052 896	10.8	0	0	39	14–78	100.0
2009–10	40 441 911	2 683 143	6.6	0	0	48	17–101	100.0
2010–11	40 902 266	1 740 035	4.3	0	0	56	21–117	100.0
2011–12	37 894 023	2 104 731	5.6	3	0.001	52	20–116	100.0
2012–13	32 508 639	801 318	2.5	0	0	35	12–75	100.0
2013–14	40 872 925	3 094 416	7.6	2	0.001	43	16–87	100.0
2014–15	39 339 271	742 291	1.9	0	0	35	13–70	100.0
2015–16	43 490 620	3 148 850	7.2	2	0.001	37	15–73	100.0

Table 8.14: Summary of observed and model-estimated total captures of white-chinned petrel by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	13	0.19	177	93–315	100.0
2003–04	120 850	6 549	5.4	18	0.27	129	72–223	100.0
2004–05	120 461	7 713	6.4	55	0.71	265	176–397	100.0
2005–06	109 951	6 619	6	72	1.09	451	282–707	100.0
2006–07	103 317	7 934	7.7	31	0.39	174	105–285	100.0
2007–08	89 536	9 050	10.1	58	0.64	270	170–431	100.0
2008–09	87 550	10 011	11.4	105	1.05	312	220–454	100.0
2009–10	92 889	9 525	10.3	73	0.77	299	195–465	100.0
2010–11	86 091	7 583	8.8	125	1.65	422	290–620	100.0
2011–12	84 424	9 336	11.1	60	0.64	203	137–305	100.0
2012–13	83 832	12 401	14.8	295	2.38	420	364–505	100.0
2013–14	85 113	13 203	15.5	153	1.16	252	208–314	100.0
2014–15	78 754	13 869	17.6	278	2	404	347–483	100.0
2015–16	78 040	14 181	18.2	160	1.13	252	211–315	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	4	0.002	137	40–410	100.0
2003–04	7 386 339	1 607 304	21.8	3	0.002	169	26–766	100.0
2004–05	3 679 965	783 812	21.3	3	0.004	38	10–119	100.0
2005–06	3 691 809	705 945	19.1	1	0.001	54	10–239	100.0
2006–07	3 740 012	1 040 948	27.8	5	0.005	31	12–75	100.0
2007–08	2 246 689	421 900	18.8	4	0.009	23	9–53	100.0
2008–09	3 114 733	937 496	30.1	3	0.003	24	8–57	100.0
2009–10	2 996 544	673 333	22.5	3	0.004	30	10–75	100.0
2010–11	3 186 899	674 572	21.2	6	0.009	31	13–70	100.0
2011–12	3 100 277	728 190	23.5	4	0.005	30	11–75	100.0
2012–13	2 876 932	560 333	19.5	1	0.002	37	11–107	100.0
2013–14	2 549 764	782 541	30.7	0	0	26	6–77	100.0
2014–15	2 412 336	725 370	30.1	2	0.003	35	8–111	100.0
2015–16	2 359 891	322 960	13.7	0	0	47	9–156	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	131	0.012	614	293–1 320	100.0
2003–04	43 243 060	5 038 339	11.7	15	0.003	355	96–1 093	100.0
2004–05	41 855 058	2 883 725	6.9	11	0.004	923	290–2 538	100.0
2005–06	37 152 933	3 802 951	10.2	13	0.003	522	130–1 809	100.0
2006–07	38 129 760	2 315 772	6.1	14	0.006	919	271–2 694	100.0
2007–08	41 512 898	3 593 511	8.7	10	0.003	744	204–2 265	100.0
2008–09	37 437 488	4 052 896	10.8	1	0	666	185–1 863	100.0
2009–10	40 441 911	2 683 143	6.6	1	0	612	155–1 869	100.0
2010–11	40 902 266	1 740 035	4.3	15	0.009	733	215–2 140	100.0
2011–12	37 894 023	2 104 731	5.6	1	0	593	138–1 997	100.0
2012–13	32 508 639	801 318	2.5	0	0	539	132–1 698	100.0
2013–14	40 872 925	3 094 416	7.6	36	0.012	726	249–1 810	100.0
2014–15	39 339 271	742 291	1.9	11	0.015	598	183–1 600	100.0
2015–16	43 490 620	3 148 850	7.2	79	0.025	742	294–1 823	100.0

Table 8.15: Summary of observed and model-estimated total captures of sooty shearwaters by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	118	1.73	996	672–1 482	100.0
2003–04	120 850	6 549	5.4	53	0.81	588	367–914	100.0
2004–05	120 461	7 713	6.4	75	0.97	615	392–953	100.0
2005–06	109 951	6 619	6	169	2.55	1 001	667–1 513	100.0
2006–07	103 317	7 934	7.7	84	1.06	589	376–910	100.0
2007–08	89 536	9 050	10.1	82	0.91	458	297–698	100.0
2008–09	87 550	10 011	11.4	153	1.53	582	406–843	100.0
2009–10	92 889	9 525	10.3	48	0.5	359	219–576	100.0
2010–11	86 091	7 583	8.8	118	1.56	536	365–791	100.0
2011–12	84 424	9 336	11.1	34	0.36	290	163–514	100.0
2012–13	83 832	12 401	14.8	135	1.09	343	240–531	100.0
2013–14	85 113	13 203	15.5	125	0.95	358	251–535	100.0
2014–15	78 754	13 869	17.6	136	0.98	380	267–576	100.0
2015–16	78 040	14 181	18.2	62	0.44	251	156–433	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	7	0.003	13	7–35	100.0
2003–04	7 386 339	1 607 304	21.8	2	0.001	6	2–20	100.0
2004–05	3 679 965	783 812	21.3	0	0	2	0–8	100.0
2005–06	3 691 809	705 945	19.1	0	0	2	0–7	100.0
2006–07	3 740 012	1 040 948	27.8	2	0.002	4	2–9	100.0
2007–08	2 246 689	421 900	18.8	0	0	1	0–4	100.0
2008–09	3 114 733	937 496	30.1	0	0	1	0–3	100.0
2009–10	2 996 544	673 333	22.5	0	0	1	0–5	100.0
2010–11	3 186 899	674 572	21.2	0	0	1	0–4	100.0
2011–12	3 100 277	728 190	23.5	0	0	0	0–3	100.0
2012–13	2 876 932	560 333	19.5	0	0	1	0–4	100.0
2013–14	2 549 764	782 541	30.7	0	0	1	0–3	100.0
2014–15	2 412 336	725 370	30.1	0	0	1	0–4	100.0
2015–16	2 359 891	322 960	13.7	0	0	1	0–6	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	25	0.002	63	39–99	100.0
2003–04	43 243 060	5 038 339	11.7	17	0.003	73	35–143	100.0
2004–05	41 855 058	2 883 725	6.9	3	0.001	44	14–108	100.0
2005–06	37 152 933	3 802 951	10.2	3	0.001	20	6–50	100.0
2006–07	38 129 760	2 315 772	6.1	1	0	32	5–95	100.0
2007–08	41 512 898	3 593 511	8.7	6	0.002	35	13–79	100.0
2008–09	37 437 488	4 052 896	10.8	0	0	36	3–105	100.0
2009–10	40 441 911	2 683 143	6.6	7	0.003	56	18–135	100.0
2010–11	40 902 266	1 740 035	4.3	0	0	32	3–99	100.0
2011–12	37 894 023	2 104 731	5.6	0	0	27	2–80	100.0
2012–13	32 508 639	801 318	2.5	0	0	26	3–79	100.0
2013–14	40 872 925	3 094 416	7.6	0	0	19	1–55	100.0
2014–15	39 339 271	742 291	1.9	0	0	19	2–56	100.0
2015–16	43 490 620	3 148 850	7.2	0	0	14	1–44	100.0

Table 8.16: Summary of observed and model-estimated total captures of black petrels by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	0	0	48	24–79	100.0
2003–04	120 850	6 549	5.4	0	0	50	26–83	100.0
2004–05	120 461	7 713	6.4	0	0	60	32–97	100.0
2005–06	109 951	6 619	6	0	0	55	29–91	100.0
2006–07	103 317	7 934	7.7	1	0.01	49	25–81	100.0
2007–08	89 536	9 050	10.1	0	0	43	21–72	100.0
2008–09	87 550	10 011	11.4	0	0	44	22–73	100.0
2009–10	92 889	9 525	10.3	0	0	47	24–77	100.0
2010–11	86 091	7 583	8.8	0	0	44	22–74	100.0
2011–12	84 424	9 336	11.1	0	0	43	21–71	100.0
2012–13	83 832	12 401	14.8	0	0	38	19–65	100.0
2013–14	85 113	13 203	15.5	5	0.04	42	23–68	100.0
2014–15	78 754	13 869	17.6	1	0.01	33	16–57	100.0
2015–16	78 040	14 181	18.2	13	0.09	43	26–66	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	2	0.001	475	164–1 374	100.0
2003–04	7 386 339	1 607 304	21.8	1	0.001	290	104–803	100.0
2004–05	3 679 965	783 812	21.3	0	0	225	67–749	100.0
2005–06	3 691 809	705 945	19.1	0	0	166	60–416	100.0
2006–07	3 740 012	1 040 948	27.8	0	0	117	44–285	100.0
2007–08	2 246 689	421 900	18.8	1	0.002	96	34–229	100.0
2008–09	3 114 733	937 496	30.1	2	0.002	120	46–292	100.0
2009–10	2 996 544	673 333	22.5	16	0.024	122	57–266	100.0
2010–11	3 186 899	674 572	21.2	1	0.001	146	57–342	100.0
2011–12	3 100 277	728 190	23.5	1	0.001	103	38–247	100.0
2012–13	2 876 932	560 333	19.5	0	0	116	41–288	100.0
2013–14	2 549 764	782 541	30.7	0	0	95	31–257	100.0
2014–15	2 412 336	725 370	30.1	0	0	68	18–197	100.0
2015–16	2 359 891	322 960	13.7	7	0.022	102	36–262	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	0	0	661	264–1 659	100.0
2003–04	43 243 060	5 038 339	11.7	2	0	608	237–1 478	100.0
2004–05	41 855 058	2 883 725	6.9	1	0	573	215–1 459	100.0
2005–06	37 152 933	3 802 951	10.2	2	0.001	568	204–1 498	100.0
2006–07	38 129 760	2 315 772	6.1	4	0.002	710	246–1 870	100.0
2007–08	41 512 898	3 593 511	8.7	3	0.001	501	187–1 249	100.0
2008–09	37 437 488	4 052 896	10.8	8	0.002	456	172–1 156	100.0
2009–10	40 441 911	2 683 143	6.6	33	0.012	548	220–1 344	100.0
2010–11	40 902 266	1 740 035	4.3	2	0.001	488	191–1 201	100.0
2011–12	37 894 023	2 104 731	5.6	0	0	436	161–1 128	100.0
2012–13	32 508 639	801 318	2.5	2	0.002	290	122–687	100.0
2013–14	40 872 925	3 094 416	7.6	7	0.002	297	121–729	100.0
2014–15	39 339 271	742 291	1.9	2	0.003	278	112–696	100.0
2015–16	43 490 620	3 148 850	7.2	0	0	216	83–539	100.0

Table 8.17: Summary of observed and model-estimated total captures of grey petrels by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	0	0	7	0–20	100.0
2003–04	120 850	6 549	5.4	2	0.03	13	4–28	100.0
2004–05	120 461	7 713	6.4	4	0.05	17	7–35	100.0
2005–06	109 951	6 619	6	1	0.02	10	2–27	100.0
2006–07	103 317	7 934	7.7	3	0.04	17	5–39	100.0
2007–08	89 536	9 050	10.1	2	0.02	11	3–25	100.0
2008–09	87 550	10 011	11.4	0	0	8	0–24	100.0
2009–10	92 889	9 525	10.3	9	0.09	35	17–70	100.0
2010–11	86 091	7 583	8.8	7	0.09	24	12–43	100.0
2011–12	84 424	9 336	11.1	1	0.01	5	1–14	100.0
2012–13	83 832	12 401	14.8	9	0.07	13	9–23	100.0
2013–14	85 113	13 203	15.5	12	0.09	17	12–28	100.0
2014–15	78 754	13 869	17.6	5	0.04	8	5–17	100.0
2015–16	78 040	14 181	18.2	3	0.02	6	3–15	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	5	0.002	73	36–129	100.0
2003–04	7 386 339	1 607 304	21.8	4	0.002	56	28–99	100.0
2004–05	3 679 965	783 812	21.3	2	0.003	36	17–69	100.0
2005–06	3 691 809	705 945	19.1	7	0.01	40	21–72	100.0
2006–07	3 740 012	1 040 948	27.8	19	0.018	51	33–82	100.0
2007–08	2 246 689	421 900	18.8	1	0.002	15	5–31	100.0
2008–09	3 114 733	937 496	30.1	6	0.006	21	11–38	100.0
2009–10	2 996 544	673 333	22.5	1	0.001	25	11–45	100.0
2010–11	3 186 899	674 572	21.2	0	0	19	7–38	100.0
2011–12	3 100 277	728 190	23.5	2	0.003	17	7–31	100.0
2012–13	2 876 932	560 333	19.5	0	0	19	8–38	100.0
2013–14	2 549 764	782 541	30.7	1	0.001	20	8–38	100.0
2014–15	2 412 336	725 370	30.1	0	0	18	7–36	100.0
2015–16	2 359 891	322 960	13.7	0	0	21	8–41	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	58	0.005	322	149–708	100.0
2003–04	43 243 060	5 038 339	11.7	0	0	195	60–525	100.0
2004–05	41 855 058	2 883 725	6.9	1	0	206	63–574	100.0
2005–06	37 152 933	3 802 951	10.2	0	0	200	56–574	100.0
2006–07	38 129 760	2 315 772	6.1	0	0	220	61–663	100.0
2007–08	41 512 898	3 593 511	8.7	4	0.001	232	68–661	100.0
2008–09	37 437 488	4 052 896	10.8	3	0.001	192	52–602	100.0
2009–10	40 441 911	2 683 143	6.6	0	0	253	66–782	100.0
2010–11	40 902 266	1 740 035	4.3	0	0	232	68–669	100.0
2011–12	37 894 023	2 104 731	5.6	0	0	185	54–543	100.0
2012–13	32 508 639	801 318	2.5	0	0	169	49–477	100.0
2013–14	40 872 925	3 094 416	7.6	2	0.001	188	58–538	100.0
2014–15	39 339 271	742 291	1.9	3	0.004	208	62–588	100.0
2015–16	43 490 620	3 148 850	7.2	0	0	170	47–489	100.0

Table 8.18: Summary of observed and model-estimated total captures of flesh-footed shearwaters by October fishing year in trawl (effort in tows), surface-longline (effort in hooks) and bottom-longline (effort in hooks) fisheries between 2002–03 and 2015–16. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. Caps, observed captures; % obs, percentage of effort observed; % incl, percentage of total effort included in the model. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Year	Fishing effort			Seabirds		Model estimates		
	All effort	Observed	% obs	Caps	Rate	Mean	95% c.i.	% incl
2002–03	130 176	6 840	5.3	0	0	132	69–248	100.0
2003–04	120 850	6 549	5.4	0	0	122	65–216	100.0
2004–05	120 461	7 713	6.4	0	0	122	66–209	100.0
2005–06	109 951	6 619	6	8	0.12	115	67–188	100.0
2006–07	103 317	7 934	7.7	6	0.08	115	65–201	100.0
2007–08	89 536	9 050	10.1	6	0.07	103	58–177	100.0
2008–09	87 550	10 011	11.4	3	0.03	102	56–170	100.0
2009–10	92 889	9 525	10.3	2	0.02	121	64–225	100.0
2010–11	86 091	7 583	8.8	15	0.2	123	71–214	100.0
2011–12	84 424	9 336	11.1	1	0.01	95	51–167	100.0
2012–13	83 832	12 401	14.8	0	0	106	56–201	100.0
2013–14	85 113	13 203	15.5	9	0.07	103	56–189	100.0
2014–15	78 754	13 869	17.6	9	0.06	95	53–166	100.0
2015–16	78 040	14 181	18.2	2	0.01	89	45–168	100.0
Surface longline								
2002–03	10 771 038	2 195 152	20.4	0	0	671	303–1 341	100.0
2003–04	7 386 339	1 607 304	21.8	0	0	432	192–894	100.0
2004–05	3 679 965	783 812	21.3	1	0.001	252	94–569	100.0
2005–06	3 691 809	705 945	19.1	4	0.006	256	106–551	100.0
2006–07	3 740 012	1 040 948	27.8	3	0.003	207	81–462	100.0
2007–08	2 246 689	421 900	18.8	2	0.005	172	64–392	100.0
2008–09	3 114 733	937 496	30.1	0	0	220	84–491	100.0
2009–10	2 996 544	673 333	22.5	0	0	216	85–464	100.0
2010–11	3 186 899	674 572	21.2	2	0.003	275	108–619	100.0
2011–12	3 100 277	728 190	23.5	0	0	217	80–494	100.0
2012–13	2 876 932	560 333	19.5	0	0	199	72–467	100.0
2013–14	2 549 764	782 541	30.7	0	0	170	54–402	100.0
2014–15	2 412 336	725 370	30.1	1	0.001	103	27–281	100.0
2015–16	2 359 891	322 960	13.7	0	0	143	46–349	100.0
Bottom longline								
2002–03	37 793 868	10 774 720	28.5	0	0	521	290–1 029	100.0
2003–04	43 243 060	5 038 339	11.7	3	0.001	460	254–960	100.0
2004–05	41 855 058	2 883 725	6.9	9	0.003	436	243–856	100.0
2005–06	37 152 933	3 802 951	10.2	0	0	344	184–684	100.0
2006–07	38 129 760	2 315 772	6.1	0	0	356	193–710	100.0
2007–08	41 512 898	3 593 511	8.7	0	0	300	168–577	100.0
2008–09	37 437 488	4 052 896	10.8	15	0.004	318	183–603	100.0
2009–10	40 441 911	2 683 143	6.6	14	0.005	290	167–548	100.0
2010–11	40 902 266	1 740 035	4.3	0	0	332	186–623	100.0
2011–12	37 894 023	2 104 731	5.6	0	0	281	164–499	100.0
2012–13	32 508 639	801 318	2.5	2	0.002	306	170–581	100.0
2013–14	40 872 925	3 094 416	7.6	31	0.01	308	180–592	100.0
2014–15	39 339 271	742 291	1.9	8	0.011	287	158–549	100.0
2015–16	43 490 620	3 148 850	7.2	12	0.004	237	134–440	100.0

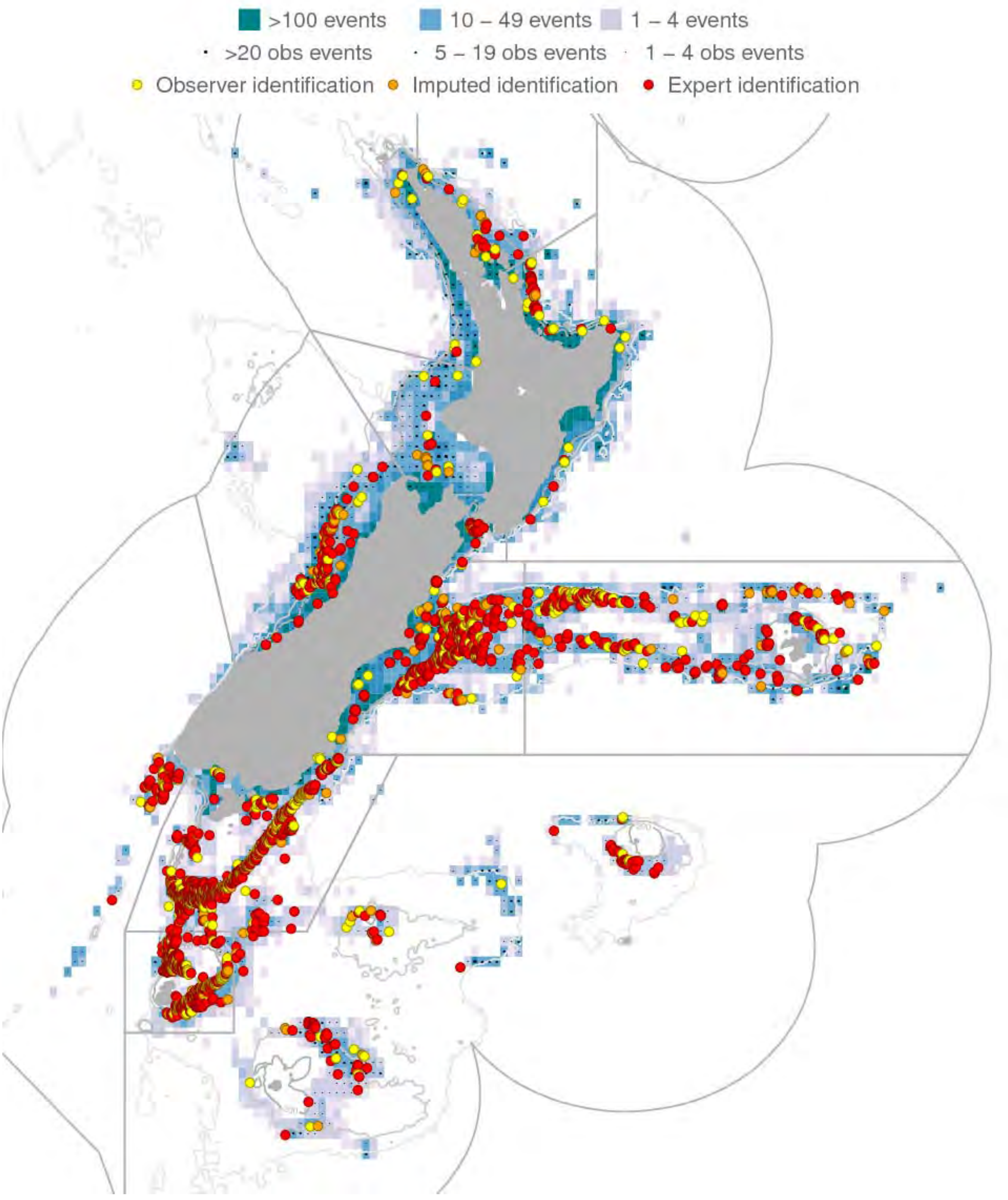


Figure 8.7: Map of trawl fishing effort and all observed seabird captures in trawls, October 2002 to September 2016. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. 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. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Table 8.19: Summary of seabirds observed captured in trawl fisheries 2002–03 to 2015–16. Declared target species are: SQU, arrow squid; HOK+, hoki, hake, ling; MID, other middle-depth species – silver, white, and common warehou, barracouta, alfonsinos, stargazer, redbait, rubyfish; SCI, scampi; ORH+, orange roughy and oreos; SBW, southern blue whiting; JMA, Jack mackerels; INS, other inshore species for which one or more captures have been observed – elephantfish, gemfish, gurnard, tarakihi, red cod, spiny dogfish, John dory, snapper; FLA, flatfishes. http://data.dragonfly.co.nz/psc. Data version v2017v1. [Continued on next page]

Species name or group	Declared target species								
	SQU	HOK+	MID	SCI	JMA	INS	SBW	ORH+	FLA	ALL
White-capped albatross	974	119	139	27	16	31	0	4	4	1 287
Salvin's albatross	23	189	134	44	2	25	16	19	0	408
Southern Buller's albatross	154	124	58	9	6	1	1	5	0	349
Albatrosses	5	7	6	4	0	0	1	1	0	20
Campbell black-browed albatross	1	10	2	1	0	0	3	0	0	16
Chatham Island albatross	0	2	3	1	0	0	0	9	0	14
Southern royal albatross	6	1	2	0	0	0	1	1	0	11
Smaller albatrosses	0	2	2	2	0	0	1	0	0	5
Black-browed albatross	1	2	0	0	0	1	0	0	0	4
Great albatrosses	0	2	0	0	0	0	0	0	0	2
Wandering albatrosses	0	1	1	0	0	0	0	0	0	2
Buller's 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
Royal albatrosses	1	0	0	0	0	0	0	0	0	1
Northern royal albatross	0	0	0	0	0	0	0	1	0	1
Gibson's albatross	0	0	0	0	0	0	0	1	0	1
Antipodean albatross	0	0	0	0	0	0	0	1	0	1
All albatrosses	1 166	460	347	88	24	58	23	42	4	2 124
White-chinned petrel	1 128	116	220	75	28	2	0	2	0	1 571
Sooty shearwater	822	317	239	37	9	2	0	3	0	1 429
Flesh-footed shearwater	0	4	37	37	0	20	0	0	0	98
Grey petrel	1	6	0	0	0	0	49	2	0	58
Cape petrel	2	30	3	3	0	0	1	9	0	48
Antarctic prion	34	0	0	0	0	0	0	0	0	34
Spotted shag	0	0	0	0	0	0	0	0	32	32
Common diving petrel	9	7	6	0	2	3	1	2	0	30
Westland petrel	0	23	2	0	1	2	0	0	0	28
Black petrel	0	0	0	0	0	20	0	0	0	20
Albatrosses	5	7	6	0	0	0	1	1	0	20
Fairy prion	2	6	1	0	8	0	0	0	0	17
Snares Cape petrel	0	10	0	0	1	0	3	0	0	14
Northern giant petrel	0	8	2	1	0	0	0	1	0	12
Grey-backed storm petrel	3	2	2	0	0	0	3	0	0	10
Mid-sized petrels & shearwaters	9	0	0	0	0	0	0	0	0	9
Fulmar prion	0	0	0	0	9	0	0	0	0	9
Giant petrels	7	0	1	0	0	0	0	0	0	8
New Zealand white-faced storm petrel	1	0	0	0	3	1	0	3	0	8
Short-tailed shearwater	0	3	2	0	0	0	0	1	0	6
Cape petrels	0	4	1	0	1	0	0	0	0	6
Shearwaters	0	0	0	0	0	4	0	0	0	4
Large seabirds	0	1	0	0	1	1	1	0	0	4

Table 8.19 [Continued]:

Species name or group	Declared target species									
	SQU	HOK+	MID	SCI	JMA	INS	SBW	ORH+	FLA	ALL
Petrels, prions, and shearwaters	0	0	0	0	0	1	2	1	0	4
Prions	1	2	1	0	0	0	0	0	0	4
Grey-faced petrel	0	0	0	0	0	3	0	0	0	3
Black-bellied storm petrel	1	1	0	0	0	0	0	0	0	2
Fulmars, petrels, prions and shearwaters	0	0	2	2	0	0	0	0	0	4
Broad-billed prion	0	0	2	0	0	0	0	0	0	2
Small seabirds	0	2	0	0	0	0	0	0	0	2
Gulls	0	0	0	0	0	2	0	0	0	2
Gadfly petrels	1	0	0	0	0	0	0	0	0	1
Seabirds	1	0	0	0	0	0	0	0	0	1
Storm petrels	0	1	0	0	0	0	0	0	0	1
Buller's albatross	1	0	0	0	0	0	0	0	0	1
White-headed petrel	1	0	0	0	0	0	0	0	0	1
Royal albatrosses	1	0	0	0	0	0	0	0	0	1
Buller's shearwater	0	0	0	0	0	1	0	0	0	1
Eurasian blackbird	0	0	1	0	0	0	0	0	0	1
Australasian gannet	0	0	0	0	1	0	0	0	0	1
All other birds	2 030	550	528	155	64	62	61	25	32	3 507
Grand total	3 196	1 010	875	243	88	120	84	67	36	5 631

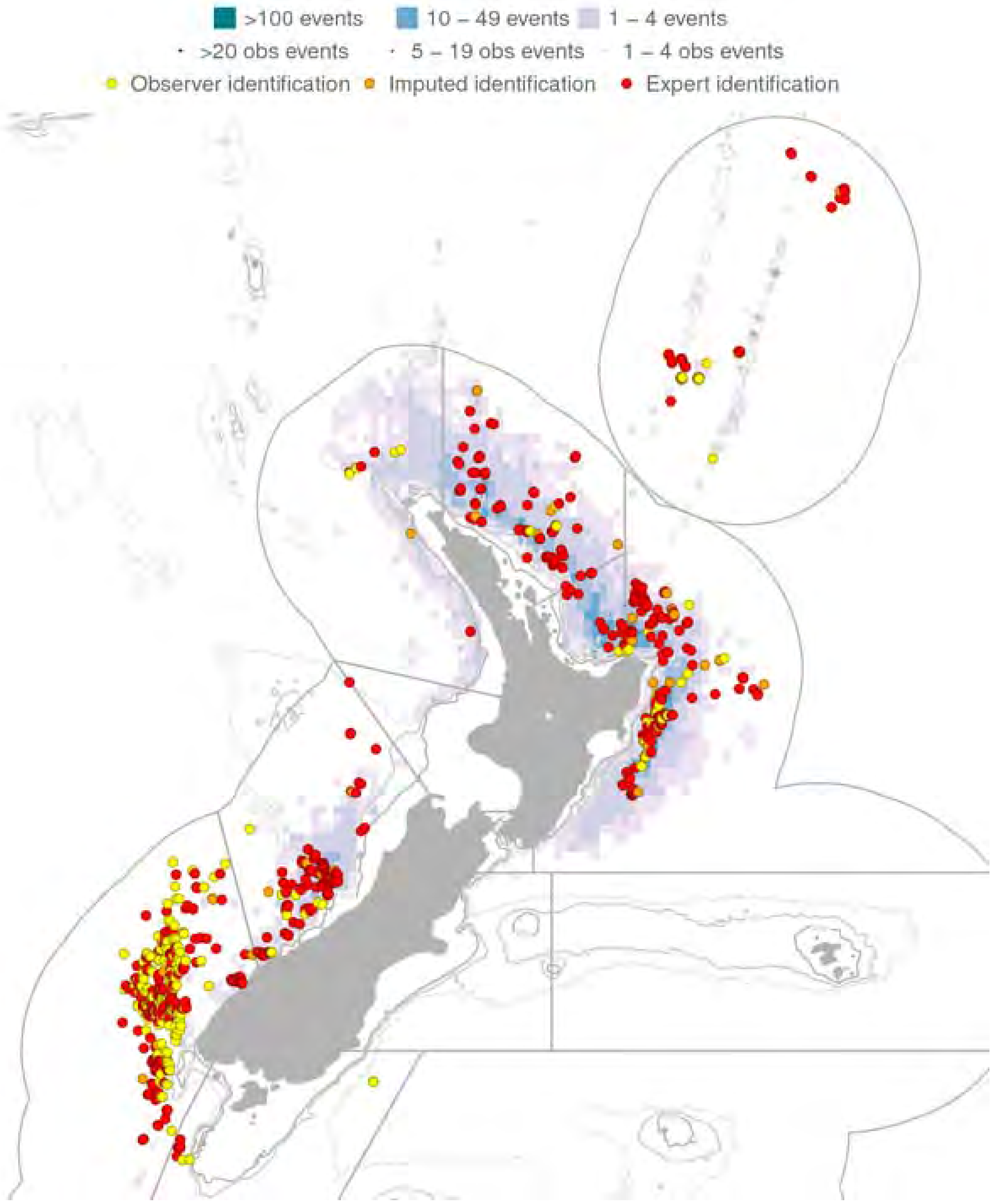


Figure 8.8: Map of surface-longline fishing effort and all observed seabird captures by surface longlines, October 2002 to September 2016. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. 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, 89.6% of effort is displayed). http://data.dragonfly.co.nz/psc. Data version v2017v1.

Table 8.20: Summary of seabirds observed captured in surface-longline fisheries 2002–03 to 2015–16. Declared target species are: SBT, southern bluefin tuna; BIG, bigeye tuna; SWO, broadbill swordfish; ALB, albacore tuna. http://data.dragonfly.co.nz/psc. Data version v2017v1.

Species name or group	Declared target species				
	SBT	SWO	BIG	ALB	ALL
Southern Buller’s albatross	431	1	12	8	452
New Zealand white-capped albatross	161	4	1	0	166
Campbell black-browed albatross	29	3	3	18	53
Gibson’s albatross	11	12	12	7	42
Antipodean albatross	7	15	11	3	36
Albatrosses	0	33	1	0	34
Wandering albatross	9	2	2	0	13
Southern royal albatross	7	0	3	0	10
Salvin’s albatross	5	0	3	0	8
Black-browed albatross	5	2	1	0	8
Antipodean and Gibson’s albatrosses	0	5	2	0	7
Light-mantled sooty albatross	2	0	0	0	2
Great albatrosses	1	0	0	0	1
Grey-headed albatross	1	0	0	0	1
Northern Buller’s albatross	1	0	0	0	1
Northern royal albatross	0	0	1	0	1
Smaller albatrosses	1	0	0	0	1
All albatross	671	77	52	36	836
Grey petrel	41	2	0	5	48
White-chinned petrel	24	7	6	2	39
Black petrel	0	3	26	2	31
Grey-faced petrel	1	3	3	16	23
Westland petrel	20	1	0	3	24
Flesh-footed shearwater	0	2	11	0	13
Sooty shearwater	3	1	0	7	11
Large seabirds	3	0	0	0	3
Cape petrels	2	0	0	0	2
Southern giant petrel	2	0	0	0	2
White-headed petrel	0	0	0	2	2
All other birds	96	19	46	37	198
Grand total	767	96	98	73	1 034

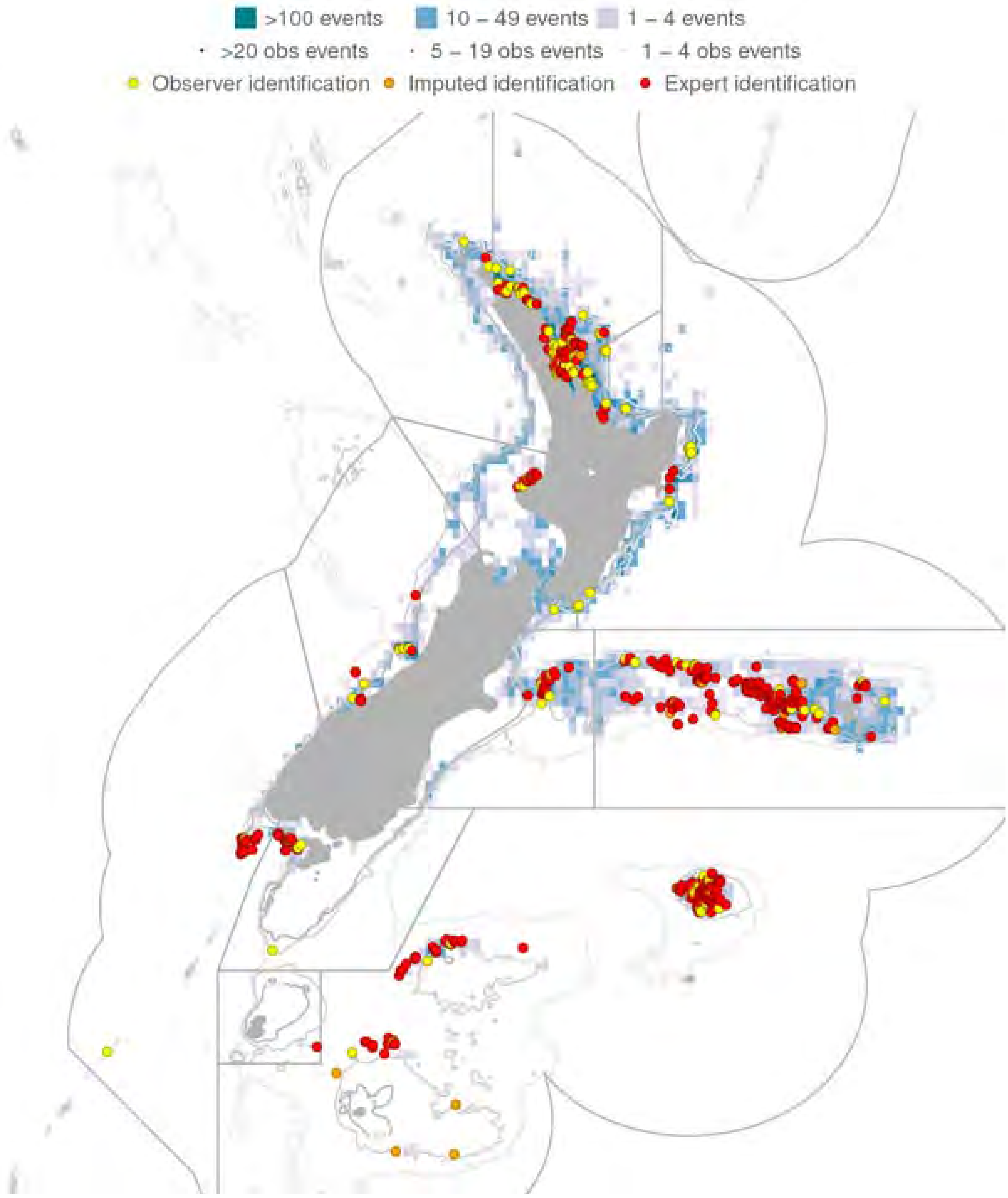


Figure 8.9: Map of bottom-longline fishing effort and all observed seabird captures by bottom longlines, October 2002 to September 2016. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by red dots. 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). http://data.dragonfly.co.nz/psc. Data version 2017v1.

Table 8.21: Summary of seabirds observed captured in bottom-longline fisheries 2002–03 to 2015–16. Declared target species are: LIN, ling; SNA, snapper; BNS, bluenose; HPB, hapuku or bass; Other – gurnard, kahawai, toothfish, ribald, school shark, tarahiki. http://data.dragonfly.co.nz/psc. Data version v2017001.

Species name or group	Declared target species					
	LIN	SNA	BNS	HPB	Other	ALL
Salvin's albatross	59	0	0	0	2	61
Chatham Island albatross	19	0	0	0	0	19
Southern Buller's albatross	11	0	3	0	0	14
White-capped albatross	5	0	0	0	0	5
Southern royal albatross	5	0	0	0	0	5
Campbell black-browed albatross	0	0	2	1	1	4
Wandering albatrosses	2	0	1	0	0	4
Albatrosses	3	0	0	0	0	3
Black-browed albatross	1	0	0	0	0	1
Indian Ocean yellow-nosed albatross	1	0	0	0	0	1
All albatross	**106**	**0**	**6**	**1**	**3**	**117**
White-chinned petrel	319	0	2	0	17	338
Flesh-footed shearwater	0	72	0	3	19	94
Grey petrel	70	1	0	0	0	71
Black petrel	0	37	23	6	0	66
Sooty shearwater	61	0	0	1	0	62
Cape petrels	23	0	0	0	0	23
Fulmars, petrels, prions and shearwaters	0	10	0	0	0	10
Southern black-backed gull	0	3	0	0	5	8
Fluttering shearwater	0	5	0	0	2	7
Common diving petrel	7	0	0	0	0	7
Prions	6	0	0	0	0	6
Grey-faced petrel	0	0	0	6	0	6
Northern giant petrel	3	1	0	0	2	6
Small seabirds	3	0	0	0	0	3
Westland petrel	1	0	0	0	2	3
Australasian gannet	0	2	0	0	0	2
Red-billed gull	0	2	0	0	0	2
Storm petrels	2	0	0	0	0	2
Seagulls	2	0	0	0	0	2
Crested penguins	1	0	0	0	0	1
Short-tailed shearwater	1	0	0	0	0	1
All other seabirds	**499**	**133**	**25**	**16**	**47**	**720**
Grand Total	**605**	**133**	**31**	**17**	**50**	**837**

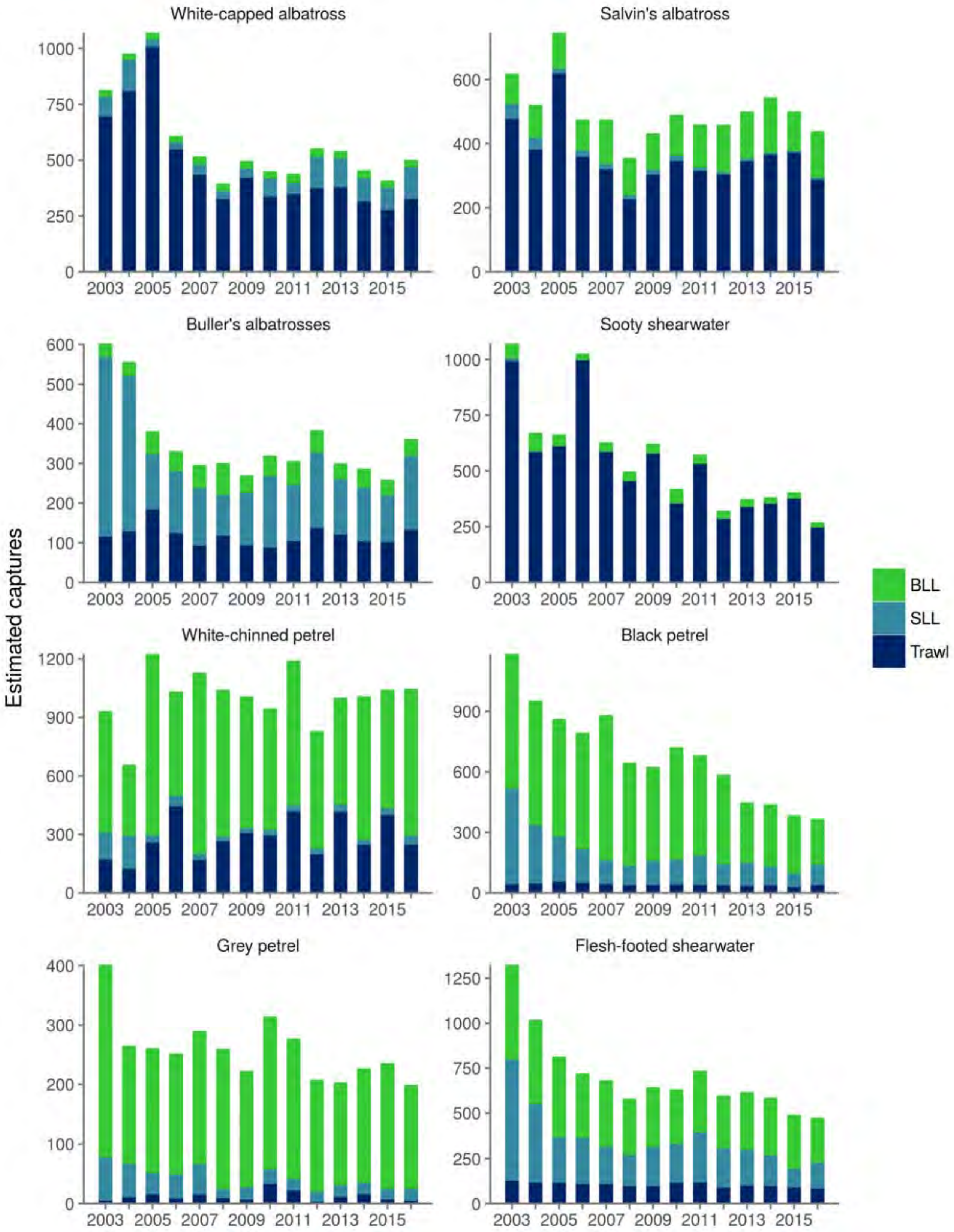


Figure 8.10: Model-based estimates of captures of the most numerous seabird taxa observed captured in trawl, surface-longline, and bottom-longline fisheries between 2002–03 and 2015–16. For confidence limits see Tables 8.10–8.15. Note that this level of aggregation conceals any different trends within a fishing method (e.g., deepwater vs. inshore and flatfish trawl, or large vs. small longliners). http://data.dragonfly.co.nz/psc. Data version v2017v1.

Over the 2002–03 to 2015–16 period, there appear to have been downward trends (across all fisheries) in the estimated captures of white-capped albatross, and non-albatross taxa other than white-chinned petrel (Figure 8.10). Estimated captures of other albatrosses and 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 catch rate indices by fishery. These are typically reported as the number of birds captured per 100 trawl tows or per 1000 longline hooks (Figure 8.11 to Figure 8.15).

For white-capped albatross, captures rates in the major offshore trawl fisheries for squid declined after 2004–05 (Figure 8.11) but showed no trend for inshore and middle depth trawlers, and increasing capture rates to a peak in 2015–16 for surface longliners targeting southern bluefin tuna. Together, these fisheries account for 76% of all estimated captures of white-capped albatross in these years.

For Salvin's albatross, captures rates have fluctuated without trend or increased in all fisheries taking substantial numbers of this species between 2002–03 and 2015–16, especially after 2006–07 (Figure 8.12). Capture rates were unusually high in the hoki trawl fishery in 2004–05. Together, these fisheries account for 69% of all estimated captures of Salvin's albatross in these years.

For Buller's albatross, estimated captures decreased in bigeye tuna target surface-longline fisheries between 2002–03 and 2015–16, while capture rates fluctuated. Captures and capture rates fluctuated with no trend in southern bluefin tuna target fisheries. The squid fishery shows some signs of an increasing trend, although had two relatively low capture rates in years 2012–13 and 2014–15 before a peak in 2015–16. The hoki trawl fishery in recent years has had higher captures and capture rates (Figure 8.13). Together these fisheries account for 65% of all estimated captures of southern Buller's albatross in these years.

For white-chinned petrel, captures rates increased between 2002–03 and 2015–16 in squid trawlers but showed little trend for bottom longliners targeting ling and bluenose, and scampi trawl. High capture rates of white-chinned petrels alternate between the squid trawl and ling longline fisheries since 2012–13. Together, these fisheries account for 77% of all estimated captures of white-chinned petrel in these years.

For sooty shearwaters, captures rates fluctuated without apparent trend between 2002–03 and 2015–16 in squid, middle-depth, hoki and scampi trawlers (Figure 8.15). Fluctuating capture rates of this species occurred in the squid and hoki trawl fisheries. Together, these fisheries account for 82% of all estimated captures of sooty shearwaters in these years.

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 (31% in trawl fisheries between 2002–03 and 2015–16, 26% in surface-longline fisheries, and 20% in bottom-longline fisheries), meaning that, all else being equal, estimates of captures may overestimate 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 across all birds or in other methods (Table 8.22). 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. in prep). 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.

Table 8.22: Percentage of observed captures that were released alive (http://data.dragonfly.co.nz/psc. Data version v2017001.)

	All birds			Albatross spp. only		
	Trawl	SLL	BLL	Trawl	SLL	BLL
2002–03	25	18	9	10	28	11
2003–04	9	30	32	4	31	80
2004–05	18	41	33	10	48	
2005–06	18	38	49	6	40	43
2006–07	18	22	12	11	23	0
2007–08	19	38	10	15	38	30
2008–09	27	26	36	20	34	50
2009–10	36	34	48	30	32	
2010–11	35	45	45	37	51	100
2011–12	25	15	70	24	17	88
2012–13	40	26	0	34	27	
2013–14	39	25	15	25	26	10
2014–15	52	42	15	46	47	100
2015–16	29	10	8	28	9	20

8.4.3 MANAGING FISHERIES INTERACTIONS

New Zealand had taken steps to reduce incidental captures of seabirds before the advent of the IPOA in 1999 and the NPOA in 2004. 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 8.23.

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 (http://www.southernseabirds.org) 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 2004 NPOA established a more comprehensive framework to reducing incidental captures approach across all fisheries (because focusing on longline fisheries like the IPOA was considered neither equitable nor sufficient).

It included two goals that set the overall direction:

1. To ensure that the long-term viability of protected seabird species is not threatened by their incidental catch in New Zealand fisheries waters or by New Zealand flagged vessels in high seas fisheries; and

2. To further reduce incidental catch of protected seabird species as far as possible, taking into account advances in technology, knowledge and financial implications.

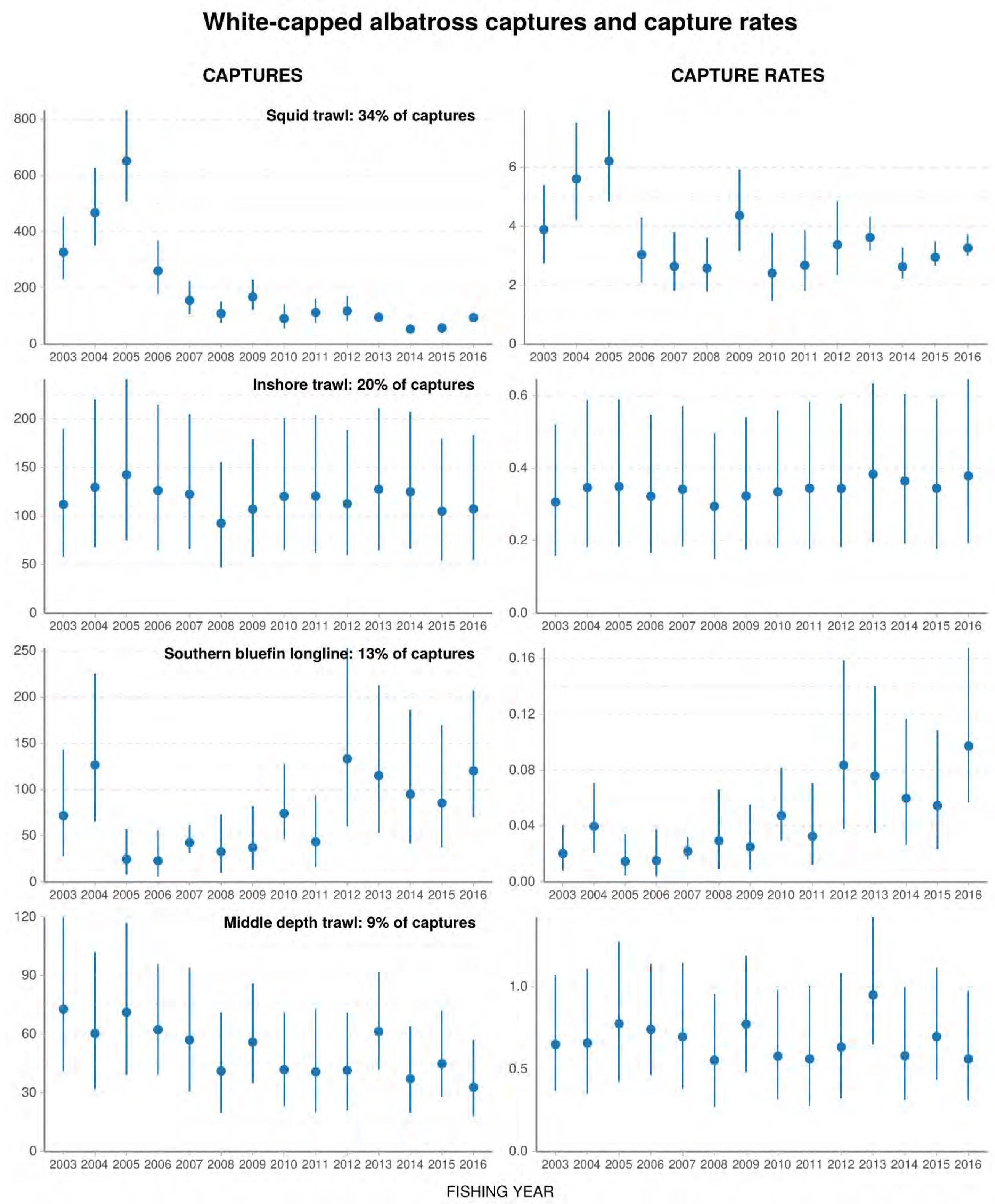


Figure 8.11: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of white-capped albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2014–15 (cumulatively, 76% of all white-capped albatross captures). http://data.dragonfly.co.nz/psc. Data version v2017001.

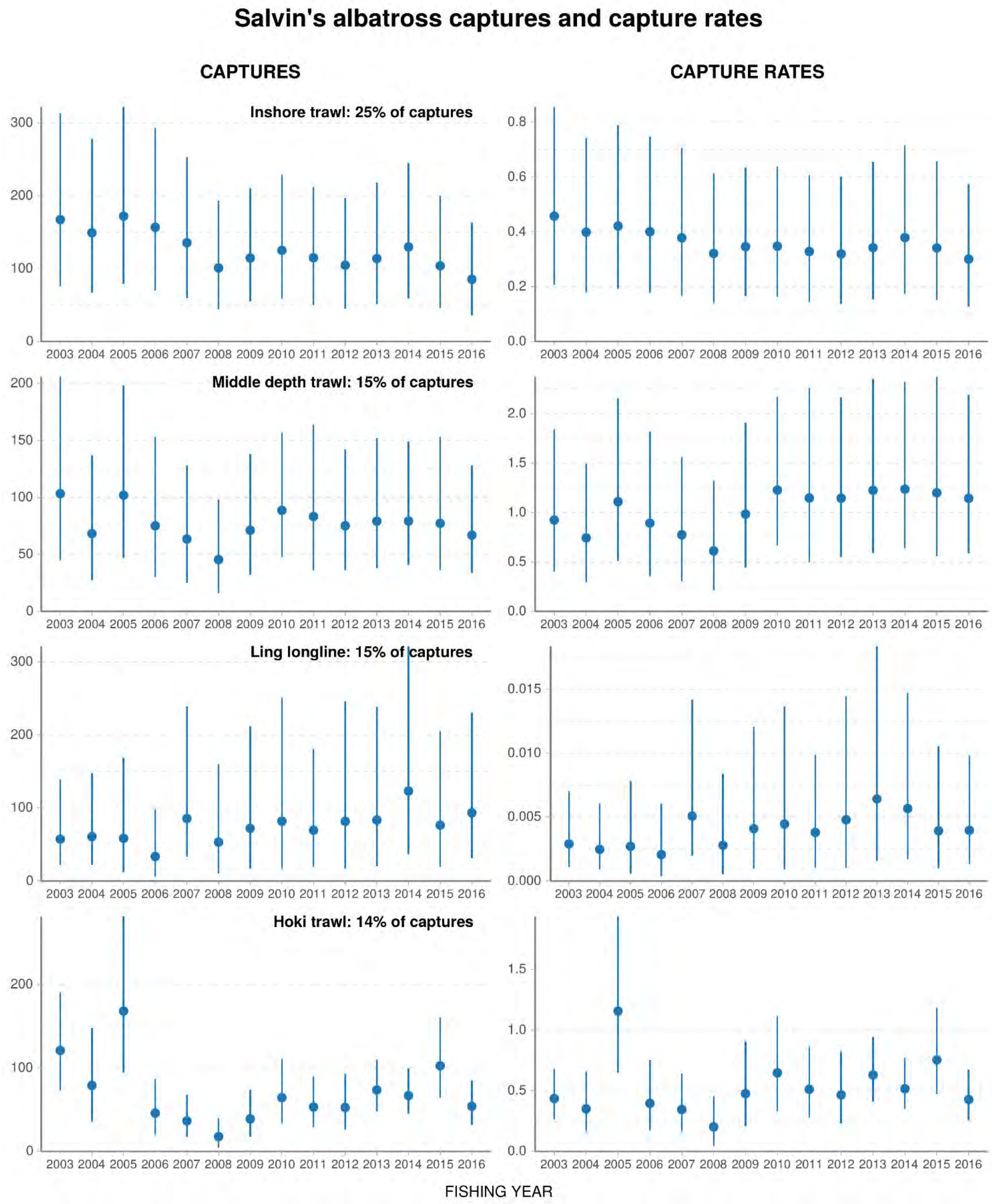


Figure 8.12: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of Salvin's albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2014–15 (cumulatively, 69% of all Salvin's albatross captures). http://data.dragonfly.co.nz/psc. Data version v2017001.

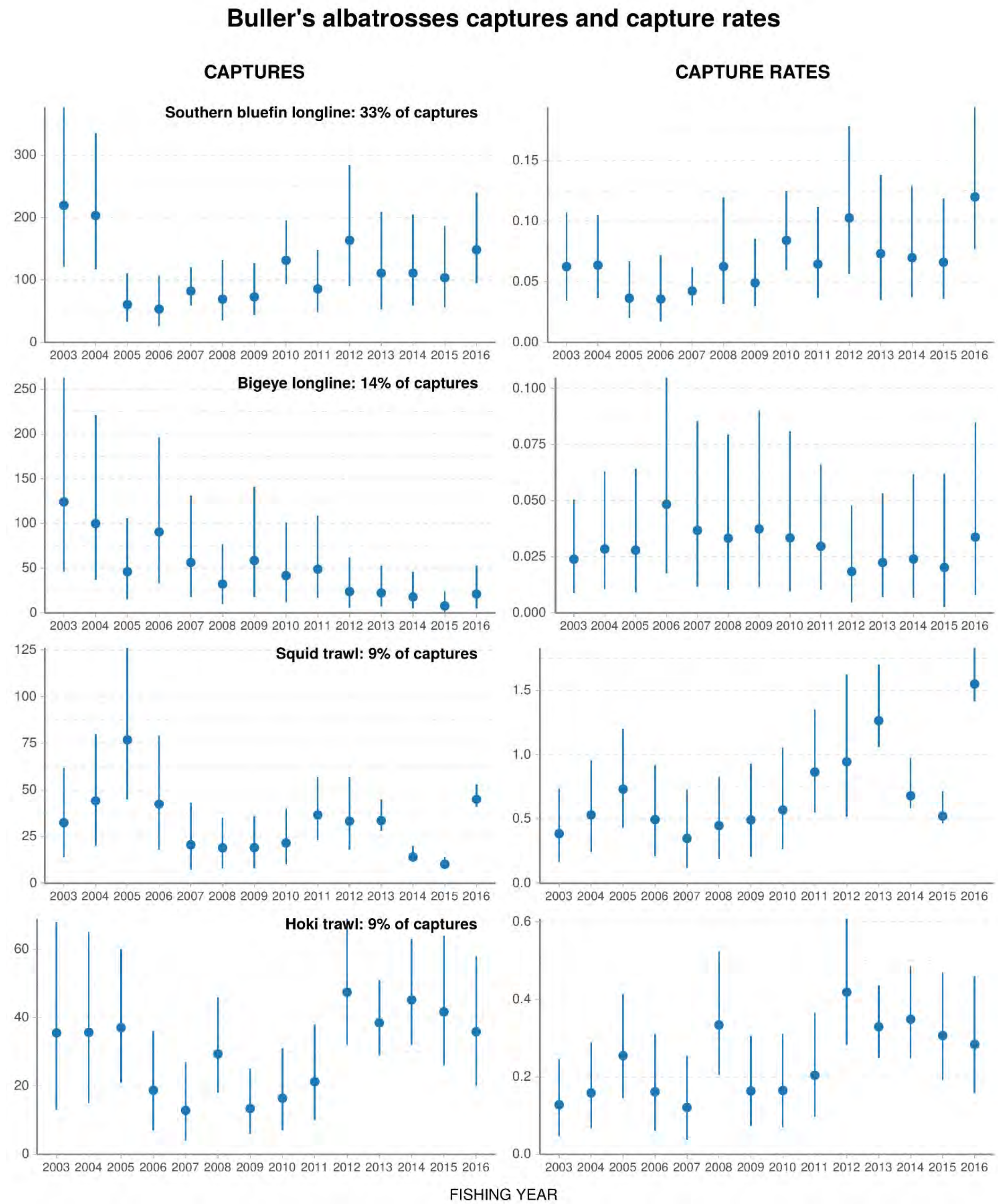


Figure 8.13: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of Buller's albatross in the four fisheries estimated to have taken the most captures between 2002–03 and 2014–15 (cumulatively, 65% of all Buller's albatross captures). http://data.dragonfly.co.nz/psc. Data version v20170001.

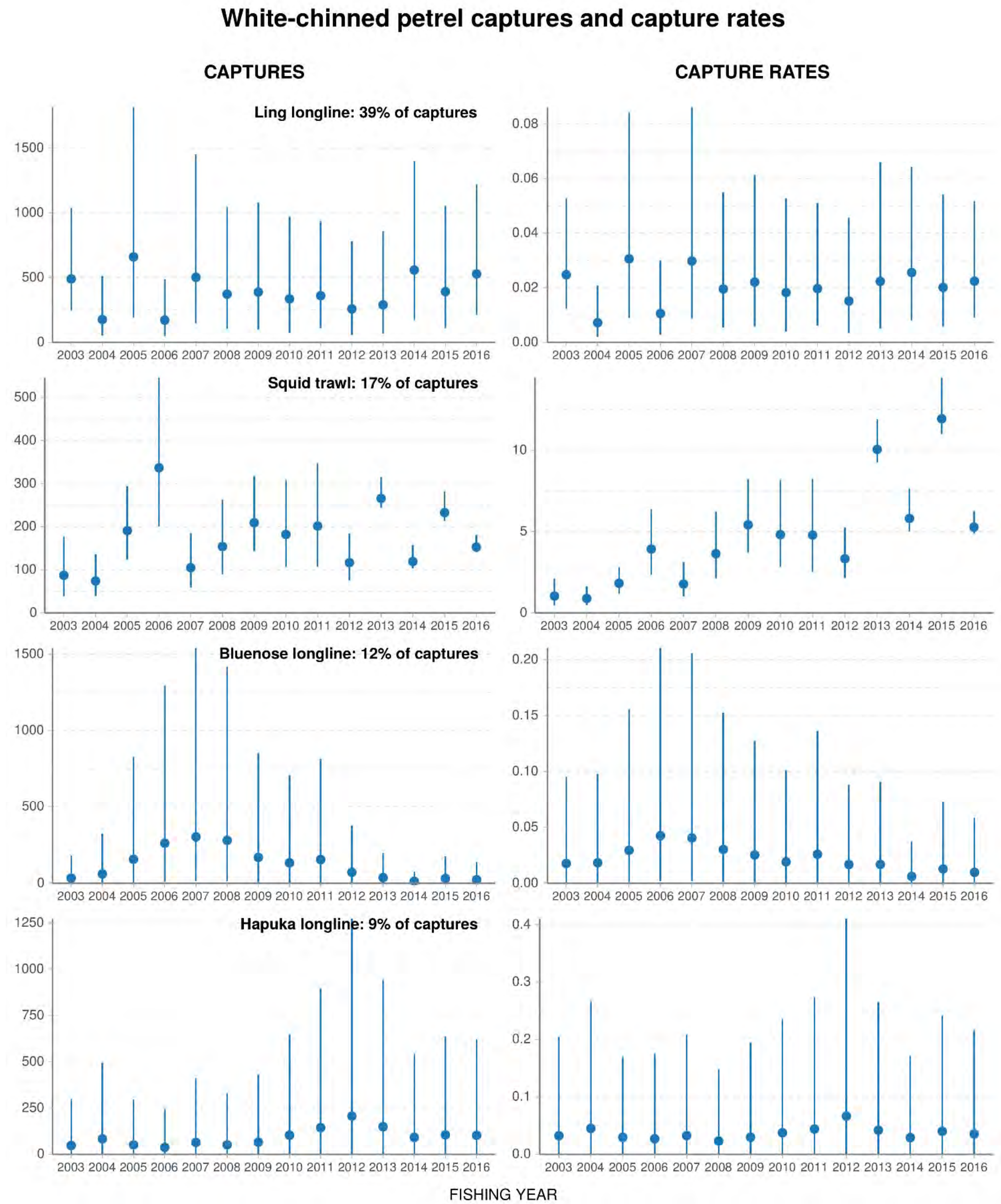


Figure 8.14: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of white-chinned petrels in the four fisheries estimated to have taken the most captures between 2002–03 and 2015–16 (cumulatively, 77% of all white-chinned petrel captures). http://data.dragonfly.co.nz/psc. Data version v20170001.

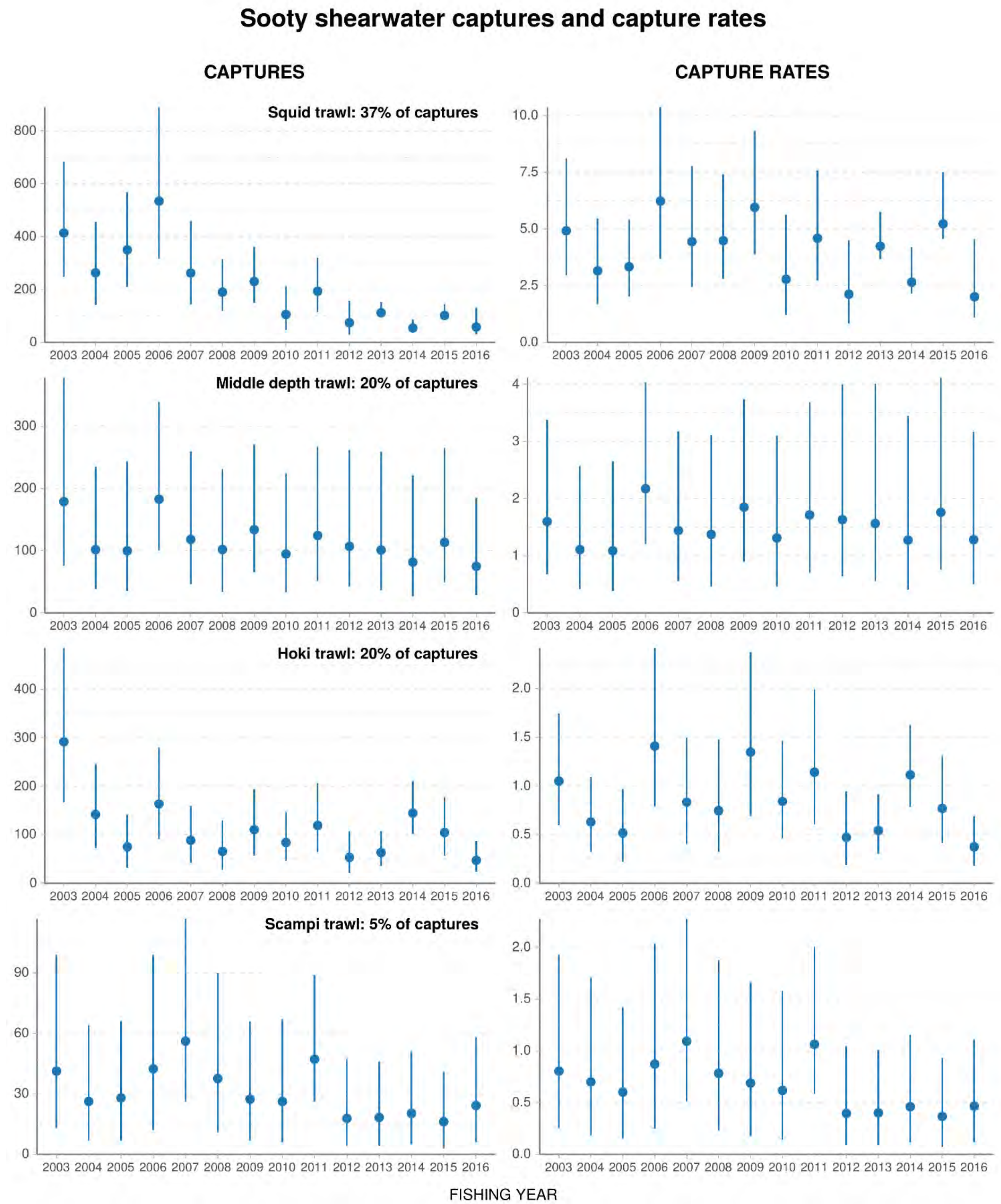


Figure 8.15: Model-based estimates of captures (left panels) and capture rates (right panels, captures per 100 trawl tows or 1000 longline hooks) of sooty shearwaters in the four fisheries estimated to have taken the most captures between 2002–03 and 2015–16 (cumulatively, 82% of all sooty shearwater captures). http://data.dragonfly.co.nz/psc. Data version v2017001.

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:

1. Implement efficient and effective management measures to achieve the goals of the NPOA, using best practice measures where possible.
2. Ensure that appropriate incentives and penalties are in place so that fishers comply with management measures.
3. 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.
4. Ensure that there is sufficient, reliable information available for the effective implementation and monitoring of management measures.
5. Establish a transparent process for monitoring progress against management measures.
6. 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.
7. Encourage and facilitate research into affected seabird species and their interactions with fisheries.
8. Encourage and facilitate research into new and innovative ways to reduce incidental catch.
9. Provide mechanisms to enable all interested parties to be involved in the reduction of incidental catch.
10. 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 seeks 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, and 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 is 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 commercial industry, recreational sector, NGOs and the relevant government departments (DOC, MPI and MFAT).

The review examines the extent to which the five-year objectives of the NPOA-seabirds have been achieved and identifies key actions as priority for the current/next NPOA-seabirds. The review will also consider whether longer-term objectives need modification, and the effectiveness of the NPOA-seabirds implementation processes.

A new NPOA-seabirds is expected to be released in 2018. The current NPOA-seabirds will remain in effect until the new NPOA-seabirds is in effect.

8.4.4 MODELLING FISHERIES INTERACTIONS AND ESTIMATING RISK

8.4.4.1 HIERARCHICAL STRUCTURE OF RISK ASSESSMENTS

Hobday et al. (2007) described a hierarchical framework for ecological risk assessment in fisheries (see Figure 8.16). 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) in order 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.

Table 8.23: (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.

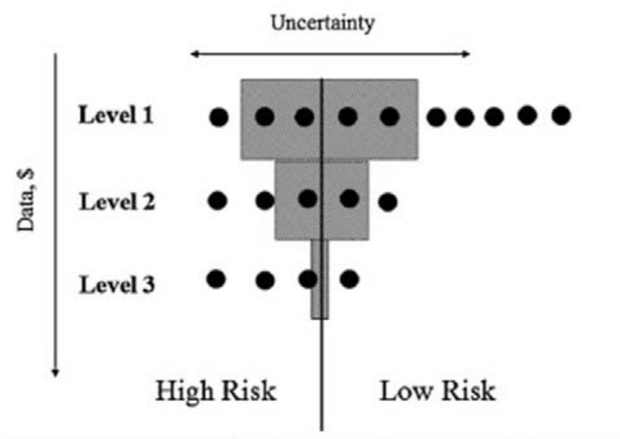


Figure 8.16: (from Hobday et al. 2007). Diagrammatic representation of the hierarchical risk assessment process where activities that present low risk are progressively screened out by assessments of increasingly high data content, sophistication and cost.

8.4.5 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 NZ 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 according to the schemas in Table 8.24 and Table 8.25 (details in 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 using the schema in Table 8.26.

Total potential and residual risk for a seabird taxon was estimated by summing the scores across all fisheries (Table 8.27) 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 8.28 shows the results for all 26 fishery groups).

Table 8.24: (modified from Fletcher 2005, Hobday et al. 2007). Exposure scores used by Rowe (2013).

Score	Descriptor	Description
0	Remote	The species will not interact directly with the fishery
1	Rare	Interactions may occur in exceptional circumstances
2	Unlikely	Evidence to suggest interactions possible
3	Possible	Evidence to suggest interactions occur, but are uncommon
4	Occasional	Interactions likely to occur on occasion
5	Likely	Interactions are expected to occur

Table 8.25: (modified from Fletcher 2005, Campbell & Gallagher 2007, Hobday et al. 2007). Consequence scores used by Rowe (2013).

Score	Descriptor	Description
1	Negligible	Some or one individual(s) impacted, no population impact
2	Minor	Some individuals are impacted, but minimal impact on population structure or dynamics. In the absence of further impact, rapid recovery would occur
3	Moderate	The level of interaction/impact is at the maximum acceptable level that still meets an objective. In the absence of further impact, recovery is expected in years
4	Major	Wider and longer-term impacts; loss of individuals; potential loss of genetic diversity. Level of impact is above the maximum acceptable level. In the absence of further impact, recovery is expected in multiple years
5	Severe	Very serious impacts occurring, loss of seabird populations causing local extinction; decline in species with single breeding population, measurable loss of genetic diversity. In the absence of further impact, recovery is expected in years to decades
6	Intolerable	Widespread and permanent/irreversible damage or loss occurring; local extinction of multiple seabird populations; serious decline of a species with a single breeding population, significant loss of genetic diversity. Even in the absence of further impact, long-term recovery period to acceptable levels will be greater than decades or may never occur

Table 8.26: (after Hobday et al. 2007). Confidence scores used by Rowe (2013).

Score	Descriptor	Rationale for confidence score
1a 1b 1c 1d	Low	Data exists, but is considered poor or conflicting. No data exists. Agreement between experts, but with low confidence. Disagreement between experts.
2a 2b 2c	High	Data exists and is considered sound. Consensus between experts. High confidence exposure to impact cannot occur (e.g., no spatial overlap of fishing activity and at-sea seabird distribution).

Table 8.27: Potential and residual risk scores for each seabird taxon with a potential risk score of 30 or more in Rowe (2013). Residual risk ('optimal risk' in Rowe 2013, not tabulated therein for grey-faced petrel or light-mantled albatross) is estimated assuming mitigation is deployed and correctly used throughout all interacting fisheries.

Taxon	Potential score	Residual score	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	–	–

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.

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.

8.4.5.1 SEABIRD SEFRA

The Spatially Explicit Fisheries Risk Assessment (SEFRA) approach used by MPI 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 MPI is a generalisation of the spatial overlap approach described by Kirby & Hobday (2007) and 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 in Sharp et al. (2011) 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 et al. 2017). The method 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 relative encounter rate of each seabird taxon with each fishery group is estimated as a function of the spatial overlap between seabird distributions (e.g., Figure 8.17) and fishing effort distributions (e.g., see Figure 8.7 to Figure 8.9). These estimates are compared with observed captures in an integrated model including all seabird groups and fisheries to estimate vulnerability (capture rates per encounter) and total captures by taxon in each fishery group. All captures are assumed fatal because of the unknown survival rate of birds released alive. Potential fatality estimates also include scalars for cryptic mortality and are subsequently compared with population estimates and biological characteristics to yield estimates of population-level risk from fishing (see method diagram in Figure 8.18).

For each taxon, the risk was assessed by dividing the estimated number of annual potential fatalities (APF) by an estimate of Potential Biological Removals (PBR, after Wade 1998). This index represents the amount of human-induced mortality a population can sustain without compromising its ability to achieve and maintain a population size above its maximum net productivity level (MNPL) or to achieve rapid recovery from a depleted state. In the risk assessment, PBR was estimated from the best available information on the demography of each taxon, including the seasonality of the distribution of various species where applicable (Figure 8.17). Because estimates of seabirds' demographic parameters and of fisheries-related mortality are imprecise, the uncertainty around the demographic and mortality estimates was propagated through the analysis. This allowed uncertainty in the resulting risk to be calculated, and also allowed the identification of parameters where improved precision would reduce overly large uncertainties. However, not all sources of uncertainty could be included, and the results are best used as a guide in the setting of management and research priorities. In general, seabird demographics, the distribution of seabirds within New Zealand waters, and sources of cryptic mortality were poorly known.

Integral to Richard & Abraham's (2013b) update of the semi-quantitative risk assessment was a simulation study (Richard & Abraham 2013a) to assess the accuracy of the approximations used in PBR calculations used by Richard et al. (2011) for seabird demographics. They showed that the PBR is typically overestimated, largely because r_{max} is overestimated by Niel & Lebreton's (2005) approximation. Richard & Abraham (2013a) therefore recommended that an additional calibration factor, ρ, be included in the calculation of the PBR to correct the approximation. The calibration factor varied between 0.17 and 0.61, depending on the seabird type; in general, the calibration factor was smaller for species with slower population growth rates, such as albatrosses, and higher for species with higher growth rates, such as shags and penguins. Previous estimates of the PBR using Niel & Lebreton's (2005) approximation for seabird populations that did not include this calibration factor are likely to have overestimated the human-caused mortalities that the populations could support (Richard & Abraham 2013a).

The management criterion used for developing the seabird risk assessment was that seabird populations should have a 95% probability of being above half the carrying capacity after 200 years, in the presence of ongoing human-caused mortalities, and environmental and demographic stochasticity (Richard & Abraham 2013b). By simulating seabird populations, the factor ρ was calculated so that this criterion would be satisfied, provided human-caused mortalities were less than the base PBR (the PBR with a recovery factor, f, of 1 and using the population size rather than a minimum population estimate). In calculation of the PBR during the simulations, the Neil & Lebreton (2005) method was used for estimating r_{max}, and the Gilbert (2009) method was used for estimating total population size. The simulations did not allow for any bias in the input parameters for individual populations.

Calculation of the PBR for a seabird species requires specification of the recovery factor, f. This factor is typically set between 0.1 and 0.5 and can be used for several purposes (e.g., Lonergan 2011). It can be used to 'protect' against errors in the input data used to calculate the PBR for individual populations, to provide for faster recovery rates, and to reflect general risk aversion (especially for endangered species). For the 2013 iteration to the risk assessment, since the derivation of uncertainty in the input parameters may already protect against potential biases, it was decided to set f = 1 for all species to assess the direct impact of fisheries, then comparable between species (Richard & Abraham 2013a, 2013b, 2015).

Table 8.28: Cumulative potential risk and residual risk scores across all seabird taxa for each fishery from Rowe (2013). Residual risk ('optimal risk' in 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 gather	0	0	0	–

Following the completion of the 2013 iteration of the Level 2 seabird risk assessment (Richard & Abraham 2013b), the Ministry for Primary Industries convened an expert workshop in November 2013 to review the Level 2 seabird risk assessment inputs and results (Walker et al. 2015). This workshop systematically reviewed input data and other available information for the 26 seabird taxa with the highest risk ratios as assessed by the Level 2 risk assessment. In summary, the results of the workshop are that:

- risk appeared to be overestimated for 14 taxa, including black petrel;
- risk appeared to be reasonably estimated for nine taxa;

- risk appeared to be underestimated for three taxa: New Zealand king shag and Gibson's and Antipodean albatrosses.

A general preponderance of overestimated risk is acceptable in a risk assessment framework so long as results are used carefully. Risk assessments are generally designed to be conservative in order to highlight gaps in information to direct future research accordingly. In contrast, any persistent significant underestimation of risk across many species is more problematic as a species may then not be subject to the additional research or management intervention required. Note however that the spatially explicit risk assessment framework is used not only to identify which species are potentially at risk, but also to inform choices about the likely effectiveness of various management options to reduce that risk, and to prioritise further research. In this context, overestimated risk scores for a particular species, fishery group, or area may lead to sub-optimal prioritisation, and ultimately delay risk reduction interventions for those species genuinely at risk. For this reason, modification to improve the Level 2 risk assessment consistent with the recommendations of this workshop was considered a high priority for all at-risk species, regardless of whether those modifications are expected to produce a decrease or an increase in overall species-level risk.

Where current risk estimates were thought to be biased in either direction, this workshop did not seek to replace or modify the existing risk estimates for each taxon, but rather gave advice on how to improve the risk assessment at the next iteration under the existing framework, and made some recommendations for further research.

The 2014 iteration of the risk assessment (as described by Richard & Abraham 2015) estimated the risk posed to each of 70 seabird taxa by trawl, longline and set-net fisheries within New Zealand's TS and EEZ. Substantial modifications to the 2013 iteration of the risk assessment (Richard & Abraham 2013b) were made in the 2014 iteration following the recommendations of the review workshop (Walker et al. 2015), these included:

- Based on their location and season, 26 captures of southern Buller's albatross were changed to be of northern Buller's albatross. One capture previously identified as Kermadec storm petrel was changed to New Zealand white-faced storm petrel. Captures identified as southern Cape petrel were removed, as only the Snares Cape petrel subspecies breeds in New Zealand.
- The population size was changed for 11 species. It was decreased for Antipodean albatross, Gibson's albatross, and Westland petrel, and increased for Salvin's albatross, New Zealand white-capped albatross, black petrel, grey petrel, flesh-footed shearwater, pied shag, Stewart Island shag, and little black shag.
- The annual survival rate was increased for New Zealand white-capped albatross, Westland petrel, black petrel, and flesh-footed shearwater.
- The proportion of adults breeding was decreased for grey petrel and New Zealand white-capped albatross, and increased for pied shag.
- The breeding season was altered for 54 species.
- The royal albatrosses were separated from the Antipodean and Gibson's albatrosses for the estimation of vulnerability was amended. The grouping of shag species, depending on whether they forage in groups or not, was also amended. The two species of Buller's albatross were also disaggregated, although identification of these species was noted to be problematic and therefore an additional source of uncertainty.
- Swordfish target surface-longline fishing was treated as a distinct fishery. The small-vessel ling bottom-longline fishery was also treated as a separate fishery.
- The at-sea distribution was changed for black petrel, Salvin's albatross, Gibson's albatross, New Zealand white-capped albatross, yellow-eyed penguin, flesh-footed shearwater, Westland petrel, New Zealand storm petrel and Kermadec storm petrel.
- A parameter was introduced to describe the proportion of birds that remain in New Zealand waters during the non-breeding season, instead of treating birds as absent during the non-breeding season.

Other changes included in the 2014 iteration of the risk assessment (Richard & Abraham 2015) were:

- The data on fishing effort and observed captures included two more years, and vulnerability was estimated using data between the 2006–07 and 2012–13 fishing years.
- The APFs were estimated on data between the 2010–11 and 2012–13 fishing years to reflect the

current level and spatial distribution of fishing effort.

The 2016 iteration of the seabird risk assessment (Richard et al. 2017) has included significant modifications to the methodology:

- instead of N_{min}, considering the total population size in the PST calculation, because uncertainty was included in the calculation, and the risk categories were defined not only from the average risk ratio but also from the upper 95% credible limit;
- using the allometric survival rate and age at first reproduction for the calculation of R_{max};
- applying a revised correction factor as the previous was found to be biologically implausible;
- applying a constraint on the fatalities calculated based on observed survival rates;
- including live release survival;
- allowing change in vulnerability over time where there is enough data;
- switching to the assumption that the number of incidents is related to vulnerability;
- Disaggregating cryptic mortality multipliers in trawl fisheries.

There were also changes made to structural assumptions and underlying data:

- the fisheries groups: inshore and flatfish trawl fisheries were merged and then split by vessel size at 17 m; middle-depth fish processing trawlers were merged, leaving freshers separate; ribald target fishing was removed from the group of other small vessel BLL and merged with the ling target fishery; and the large BLL fleet (>36m) was split based on the use of integrated weighted line (Table 8.29);
- seabird demographic data were updated, based on input from seabird experts and reviewed by the AEWG;
- the Stewart Island shag was split into two species, the Otago and Foveaux shags (Richard et al. 2017).

The 2016 iteration derives a risk ratio, which is an estimate of aggregate potential fatalities across trawl and longline fisheries relative to the Population Sustainability Threshold, PST (an analogue of the Potential Biological Removals, PBR, approach) (Richard et al. 2017).

Simulations were carried out to ensure that populations would satisfy the long-term goal of having the population above half the carrying capacity, with 95% probability. If there is environmental variability then a correction factor less than one, ρ, is required to ensure that the goal is achieved. In the 2014 iteration of the risk assessment, ρ included the correction for environmental variability, a correction to r_{max} and the correction to the population size. After splitting ρ into components, Richard et al. (2017) found that the correction required to account for environmental variability has no clear relationship with the taxonomy. For each of the simulated taxa, the value was close to 0.5.

However the 2016 iteration of the seabird risk assessment does not specifically include a recovery factor, as threat classifications often include some element of the risk posed by fishing, which would essentially double count the impacts of fishing. It would likely be preferable to conduct a more comprehensive risk assessment that compares the impacts of a range of threats on the seabird populations if sufficient information on the other risks were available. MPI has committed to further discussions around the inclusion of the recovery factor or modifications to the reference outcome in further risk assessments.

Following the changes to the risk assessment structure and inputs, the overall changes between the 2014 risk assessment and the 2016 iteration can be seen in Figure 8.19 and Table 8.30. The progressive change in risk ratio to each successive change in the risk assessment structure and input parameters is given in Table 8.31.

The combination of the use of the total population size, the allometric modelling of adult survival and age at first reproduction, and the use of different corrections for the calculation of PST, led to significant changes to the estimated risk ratio (Table 8.30 and Table 8.31). However the rankings of seabird species by risk ratio mean are similar (Figure 8.19 and Table 8.30 to Table 8.31). Even with the recovery factor not included, only one species, the black petrel, had a median risk ratio of greater than 1 or the upper 95% confidence limit higher than 2. Black petrel were classified as 'very high risk' with a risk ratio (estimated annual potential fishing-related fatalities higher than the PST) median: 1.15; 95% c.i.: 0.51–2.03 (Figure 8.19 and Table 8.30 to Table 8.31).

Seven species were classified as 'high risk' because they have a risk ratio with a median above 0.3 or with the upper

95% confidence limit above 1: Salvin's albatross (median 0.78; 95% c.i.: 0.51–1.09)., flesh-footed shearwater (2.82; 95% c.i.: 1.56–5.60), Westland petrel (0.48; 95% c.i.: 0.18–1.19), southern Buller's albatross (0.39; 95% c.i.: 0.22–0.66), Chatham Island albatross (0.36; 95% c.i.: 0.18–0.66), New Zealand white-capped albatross (0.35; 95% c.i.: 0.21–0.58), and Gibson's albatross (0.34; 95% c.i.: 0.19–0.59) (and Table 8.30 to Table 8.31) (Richard et al. 2017).

The risk ratio of five species had a median above 0.1 or the upper 95% confidence limit above 0.3 and are classified as at 'medium risk': northern Buller's albatross, Antipodean albatross, the mainland population of yellow-eyed penguin (assuming that all fisheries-related mortalities are of the mainland population); Otago shag, and northern giant petrel (Figure 8.19 and Table 8.30 to Table 8.31) (Richard et al. 2017).

In total, there were 14 400 (95% c.i.: 11 900–17 500) APF estimated across the four fishing methods (Table 8.33). The highest number of APF was in trawl fisheries, mainly of albatross, Procellaria petrels, and large shearwater species. Species with the highest levels of estimated potential fatalities in trawl fisheries were New Zealand white-capped albatross, Salvin's albatross, sooty shearwater, and white-chinned petrel, in decreasing order of mean APF (Richard et al. 2017). The target trawl fisheries that contributed the most APF were inshore (total APF 4800; 95% c.i.: 3140–7080), hoki (1540; 95% c.i.: 1140–2050), flatfish (1210; 95% c.i.: 804–1820) and middle depth (1060; 95% c.i.: 777–1410) (Table 8.33).

In bottom-longline fisheries, the species with the highest number of estimated potential fatalities in these fisheries were: white-chinned petrel, flesh-footed shearwater, Salvin's albatross and black petrel, in decreasing order of mean APF (Richard et al. 2017). The target fisheries that contributed the highest APF were: small vessels targeting ling (total APF 1090; 95% c.i.: 809–1410), and snapper (523; 95% c.i.: 385–681) (Table 8.33).

The target surface-longline fisheries that contributed the highest levels of APF were bigeye tuna (total APF 422; 95% c.i.: 318–548), small vessel southern bluefin tuna (359; 95% c.i.: 277–450), and swordfish (262; 95% c.i.: 170–372) (Table 8.33). The species with the highest number of APF in these fisheries were: white-capped albatross, northern Buller's albatross, Gibson's albatross, and black petrel, in decreasing order of mean APF (Richard et al. 2017).

Estimated APF from set-net fisheries were relatively low, (Table 8.33). Although the total estimate was low for some species, the highest number of estimated APF occurred in set-net fisheries. In particular, there were 17 (95% c.i.: 6–34) estimated APF of yellow-eyed penguin, assumed to come from the mainland population, and 21 (95% c.i.: 5–48) estimated APF of spotted shag in set-net fisheries.

Table 8.34 provides a comparison of the estimated number of annual observable captures of seabirds including and not including cryptic mortality in trawl, bottom-longline, surface-longline and set-net fisheries. Excluding cryptic mortalities, the estimated mean number of observable black petrel captures was 248 (95% c.i.: 189–318) (Richard et al. 2017).

As there have been major changes to the risk assessment methodology since the publication of the NPOA-seabirds (MPI 2013), which includes objectives based on the risk to seabirds as assessed by a previous iteration of the risk assessment (Richard & Abraham 2013b), the new base case for the 2016 iteration was run using the same dataset (2006–07 to 2010–11) used by Richard & Abraham (2013b). The results are shown in Figure 8.20 and Table 8.32. The results show a reduction in risk during the period of the NPOA for black petrel, Salvin's albatross, the mainland population of yellow-eyed penguin and spotted shag in particular. However increases to risk are evident between these periods for flesh-footed shearwater, Westland petrel and Gibson's albatross. These increases in risk can be attributed to the observation of increased captures in the later period, for example there were no observed captures of Westland petrel until 2013–14 when there were two in inshore trawl, a poorly observed fishery (Richard et al. 2017).

The method described by Richard et al. (2011), Richard & Abraham (2013b, 2015) and Richard et al. (2017) offers the following advantages that make it particularly suitable for assessing risk to multiple seabird populations from multiple fisheries:

- risk is assessed separately for each seabird taxon; fisheries managers must assess risk to seabirds with reference to units that are biologically meaningful;
- the method 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 impact estimation

at the species or subspecies level); the method can be applied to any fishery for which at least some observer data exists;

- the method does not rely on detailed population models (the necessary data for which are unavailable for the great majority of taxa) because risk is estimated as a function of population-level potential fatalities and biological parameters that are generally available from published sources;
- the method assigns risk to each taxon in an absolute sense, i.e., taxa are not merely ranked relative to one another; this allows the definition of biologically meaningful performance standards and ability to track changes in performance over time and in relation to risk management interventions;
- risk scores are quantitative and objectively scalable between fisheries or areas, so that risk at a population level can be disaggregated and assigned to different fisheries or areas based on their proportional contribution to total impact to inform risk management prioritisation;
- the method allows explicit statistical treatment of uncertainty, and does not conflate uncertainty with the resulting risk ratio; numerical inputs include error distributions and it is possible to track the propagation of uncertainty from inputs to estimates of risk; and
- the method readily incorporates new information; assumptions in the assessment are transparent and testable and, as new data becomes available, 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.

The key disadvantages of the method of Richard et al. (2011), many of which were addressed by subsequent iterations (Richard & Abraham 2013b, 2015), were that:

- fisheries for which no observer information on seabird interactions is available cannot be included in the analysis;
- the assumption that the vulnerabilities of particular seabirds to capture in different fisheries are independent does not allow ‘sharing’ of scarce observer information between fisheries within the risk assessment (addressed in subsequent iterations);
- the spatial overlap method relies on appropriate spatial and temporal scales for the distributions of birds and fishing effort being used; use of inappropriate scales can lead to misleading results (partially addressed in 2013 revision);
- strong assumptions have to be made about the distribution and productivity of some taxa, the relative vulnerability of different taxa to capture by particular fisheries, cryptic mortality associated with different fishing methods, and the applicability.

Most of these limitations are a result of the scarcity of relevant data on seabird populations and fisheries impacts and can be addressed only through the collection of more information or, in some cases, sensitivity testing. Further refinement of this method would be possible if:

- Estimates of PBR could be compared with total annual human-caused mortality rather than mortality from commercial fishing within the New Zealand region. Little is known about the impact of New Zealand recreational fishing on seabirds or fatalities in overseas fisheries of seabirds that forage beyond New Zealand’s waters.
- Better information on cryptic mortality was available. Studies on cryptic mortality are extremely limited.
- Further observer coverage was targeted at fisheries where substantial reductions in the uncertainty about potential fatalities would result (most such fisheries are poorly observed).

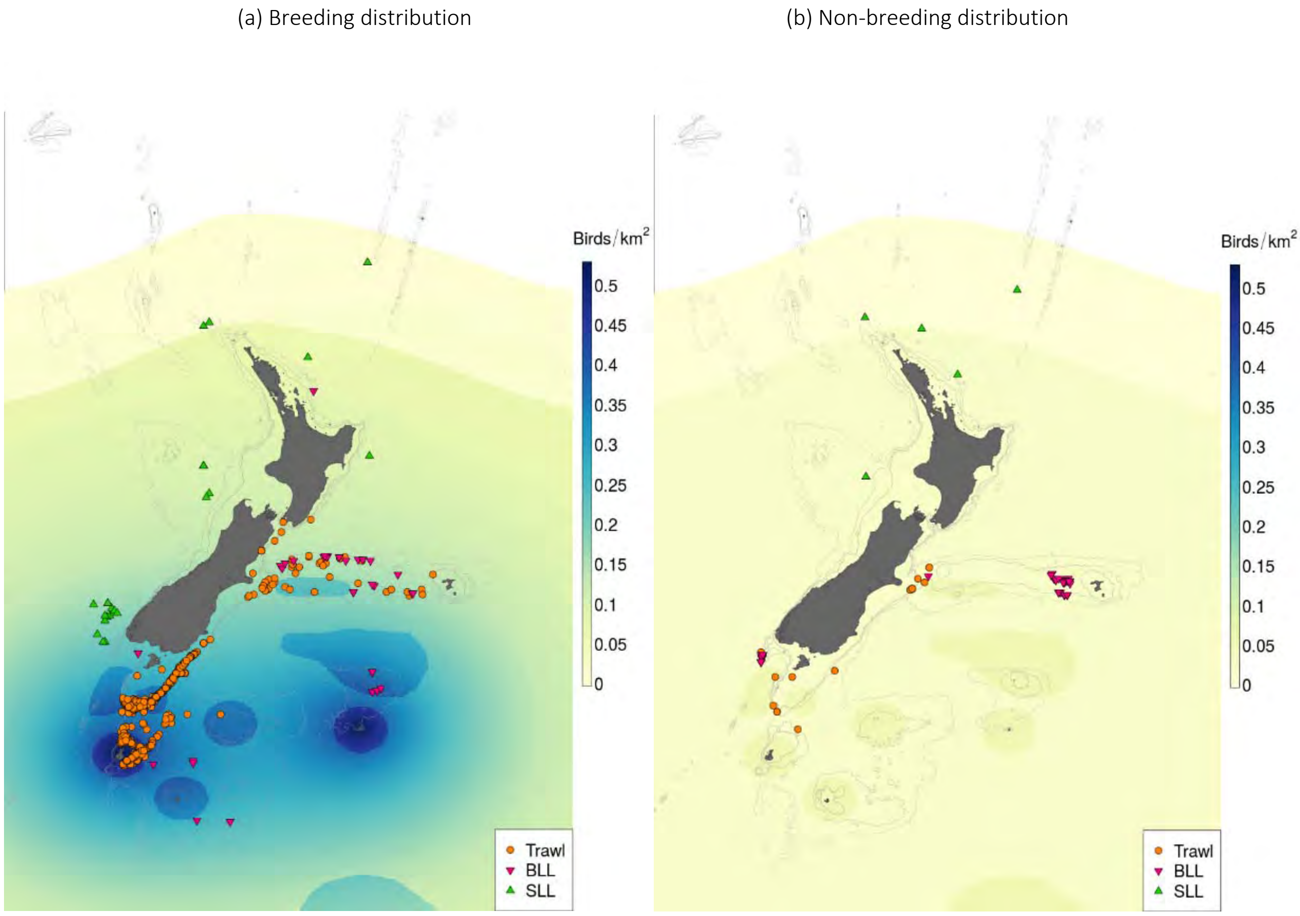


Figure 8.17: (from Richard et al. 2017, supplementary material). Relative density of white-chinned petrel. The base map for the distribution was obtained from the NABIS database. The breeding season runs from October to May. Also shown are incidental captures recorded by observers between 2006–07 and 2013–14 in trawl, surface-longline (SLL), and bottom-longline (BLL) fisheries.

Table 8.29: (from Richard et al. 2017). Revised fishery groups used in the 2016 iteration of the seabird risk assessment.

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

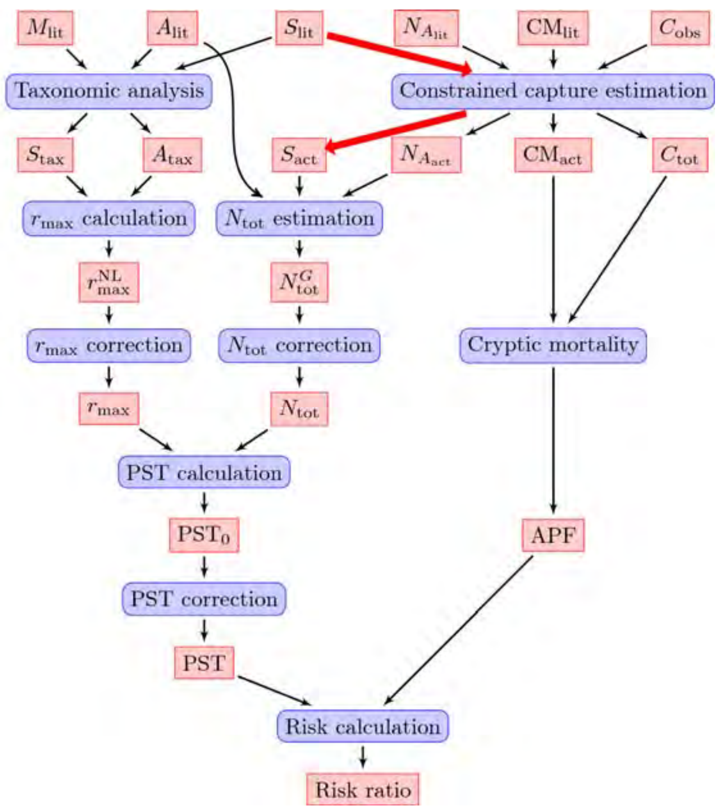


Figure 8.18: (from Richard et al. 2017). Schematic process of the estimation of risk in the seabird risk assessment after application of the considered updates to the methodology. M: body mass; A: age at first reproduction; S: adult survival rate; NA: adult population size; Ntot: total population size; CM: cryptic mortality multiplier; C: seabird captures; r_{max}: maximum net productivity rate; 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; act: corrected by the model; tot: total; NL: estimated using the approach by Niel & Lebreton (2005)'s approach; G: estimated using the formula by Gilbert (2009); 0: prior to correction.

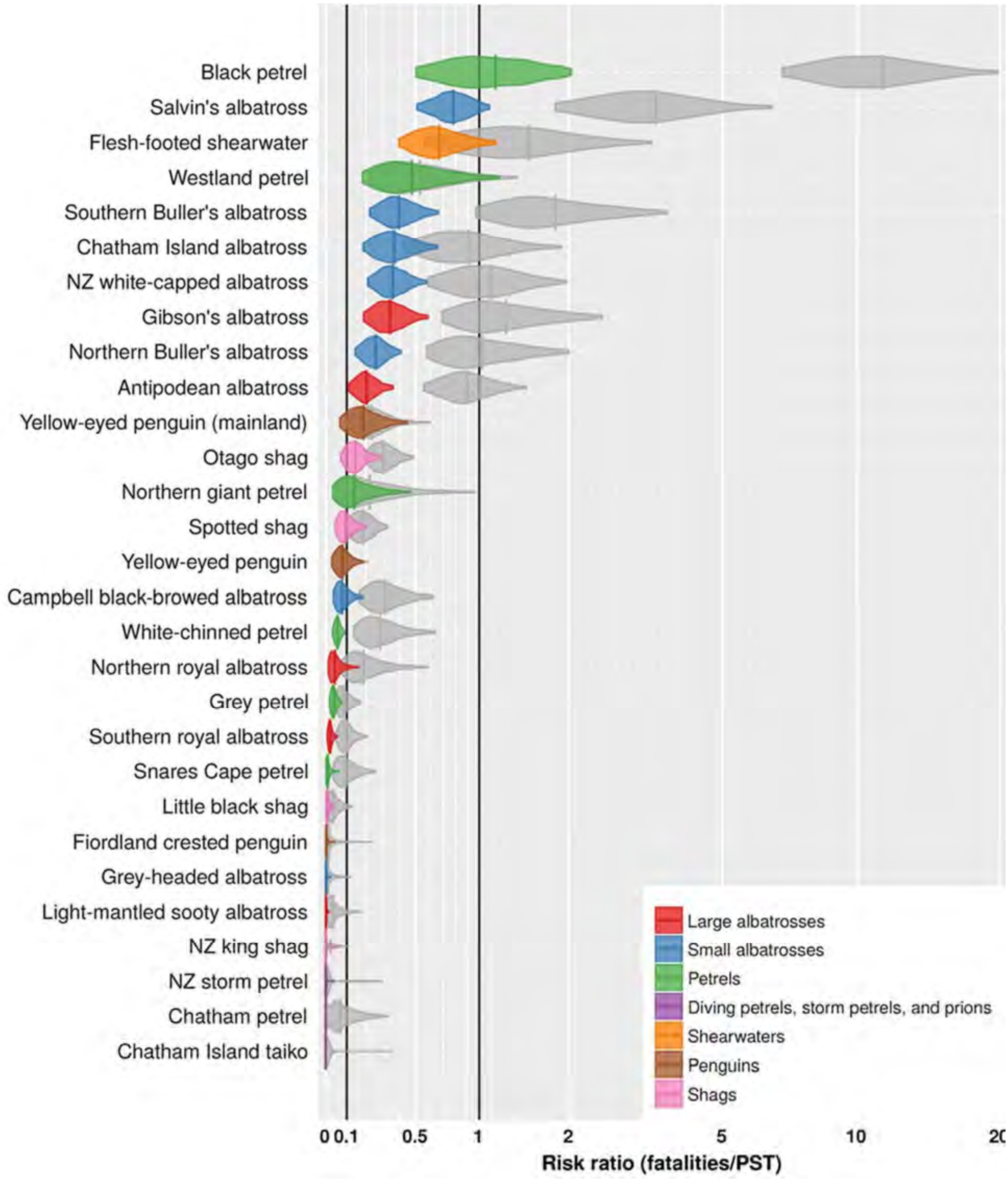


Figure 8.19: (from Richard et al. 2017). Risk ratio re-calculated on data from 2011–12 to 2014–15. The risk ratio is displayed on a logarithmic scale, with the threshold of the number of potential bird fatalities equalling the PST with f = 0:1 and f = 1 indicated by the two vertical black lines, and the distribution of the corrected risk ratios within their 95% confidence interval indicated by the coloured shapes, including the median risk ratio (vertical line). The grey shapes indicate the risk ratios from the 2014 iteration of the risk assessment (Richard & Abraham 2015). Seabird species are listed in decreasing order of the median risk ratio. Species with a risk ratio of almost zero were not included (95% upper limit with *f* = 1 less than 0.1). 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 was between 600 and 800.

Table 8.30: (from Richard et al. 2017). Comparison between the risk ratio reported by Richard & Abraham (2015), and in this study after updates. The table shows all species whose risk (in this study) had an upper 95% confidence limit greater than 0.1. Species names are coloured according to their risk category. Red: risk ratio with a median over 1 or upper 95% confidence 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.

	Before updates		After updates	
	Median	95% c.i.	Median	95% c.i.
Black petrel	**11.336**	6.853–19.814	**1.152**	0.505–2.032
Salvin's albatross	**3.440**	1.818–6.504	**0.779**	0.509–1.095
Flesh-footed shearwater	**1.500**	0.565–3.356	**0.669**	0.391–1.153
Westland petrel	**0.531**	0.214–1.375	**0.476**	0.180–1.186
Southern Buller's albatross	**1.819**	0.970–3.671	**0.392**	0.219–0.664
Chatham Island albatross	**0.906**	0.418–1.901	**0.362**	0.183–0.657
NZ white-capped albatross	**1.100**	0.587–1.968	**0.353**	0.208–0.577
Gibson's albatross	**1.256**	0.691–2.485	**0.337**	0.186–0.586
Northern Buller's albatross	**1.025**	0.579–1.998	**0.253**	0.141–0.405
Antipodean albatross	**0.894**	0.556–1.474	**0.202**	0.110–0.356
Otago shag		–	**0.144**	0.070–0.279
Northern giant petrel	**0.218**	0.046–0.963	**0.138**	0.033–0.468
Spotted shag	**0.183**	0.095–0.321	**0.093**	0.043–0.203
Campbell black-browed albatross	**0.306**	0.162–0.626	**0.077**	0.035–0.183
White-chinned petrel	**0.278**	0.136–0.641	0.055	0.032–0.089
Northern royal albatross	**0.186**	0.066–0.591	**0.043**	0.012–0.163
Stewart Island shag	**0.297**	0.186–0.485		–

Table 8.31 (from Richard et al. 2017) Progressive changes in the risk ratio. 2015 SRA results, results from the 2015 iteration; AEWG May 2016, includes use of the total population size, the allometric modelling of adult survival and age at first reproduction, and the use of different corrections for the calculation of PST; BPE from Biz Bell only, 2016 base case with black petrel population size from Bell et al. 2016; YEP latest counts, 2016 base case with latest yellow-eyed penguin counts from T. Webster and J Fyfe; Data up to 2010–11, 2016 methodology applied to the same period as the risk assessment iteration used to underpin the NPOA-seabirds 2013; New base case. For the 20 species the most at risk.

Species	2015 SRA results		AEWG May 2016		BPE from Biz Bell only		YEP latest counts		Data up to 2010-11		New base case	
	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.
Black petrel	**11.34**	6.85–19.81	**1.58**	1.05–2.12	**2.11**	1.63–2.58	**1.16**	0.50–2.05	**1.51**	0.75–2.25	**1.15**	0.51–2.03
Salvin's albatross	**3.44**	1.82–6.50	**0.79**	0.45–1.19	**0.78**	0.51–1.10	**0.78**	0.52–1.10	**0.88**	0.58–1.22	**0.78**	0.51–1.09
Flesh-footed shearwater	**1.5**	0.56–3.36	**0.55**	0.26–1.01	**0.61**	0.36–1.08	**0.67**	0.39–1.20	**0.44**	0.23–1.00	**0.67**	0.39–1.15
Southern Buller's albatross	**1.82**	0.97–3.67	**0.46**	0.23–0.82	**0.39**	0.21–0.66	**0.39**	0.21–0.66	**0.33**	0.18–0.58	**0.39**	0.22–0.66
Gibson's albatross	**1.26**	0.69–2.49	**0.46**	0.25–0.79	**0.33**	0.18–0.57	**0.34**	0.18–0.58	**0.26**	0.14–0.46	**0.34**	0.19–0.59
NZ white-capped albatross	**1.1**	0.59–1.97	**0.3**	0.15–0.60	**0.36**	0.22–0.58	**0.35**	0.21–0.57	**0.32**	0.19–0.56	**0.35**	0.21–0.58
Chatham Island albatross	**0.91**	0.42–1.90	**0.27**	0.12–0.57	**0.35**	0.18–0.66	**0.36**	0.19–0.65	**0.4**	0.20–0.72	**0.36**	0.18–0.66
Westland petrel	**0.53**	0.21–1.37	**0.29**	0.12–0.72	**0.42**	0.16–1.09	**0.49**	0.18–1.19	**0.23**	0.09–0.69	**0.48**	0.18–1.19
Northern Buller's albatross	**1.02**	0.58–2.00	**0.32**	0.17–0.54	**0.24**	0.14–0.41	**0.25**	0.14–0.40	**0.25**	0.14–0.41	**0.25**	0.14–0.41
Antipodean albatross	**0.89**	0.56–1.47	**0.34**	0.18–0.62	**0.19**	0.11–0.35	**0.2**	0.11–0.35	**0.2**	0.11–0.34	**0.2**	0.11–0.36
Yellow-eyed penguin (mainland)	**0.23**	0.10–0.60	**0.23**	0.10–0.60	**0.18**	0.06–0.43	**0.36**	0.12–0.90	**0.25**	0.09–0.62	**0.18**	0.07–0.45
Otago shag	**0.3**	0.19–0.49	**0.21**	0.10–0.42	**0.15**	0.07–0.28	**0.14**	0.07–0.28	**0.21**	0.11–0.39	**0.14**	0.07–0.28
Northern giant petrel	**0.22**	0.05–0.96	**0.08**	0.00–0.35	**0.14**	0.03–0.43	**0.14**	0.03–0.44	**0.13**	0.03–0.44	**0.14**	0.03–0.47
Spotted shag	**0.18**	0.10–0.32	**0.14**	0.06–0.31	**0.1**	0.04–0.21	**0.09**	0.04–0.20	**0.15**	0.07–0.33	**0.09**	0.04–0.20
Campbell black-browed albatross	**0.31**	0.16–0.63	**0.1**	0.04–0.24	**0.08**	0.03–0.18	**0.08**	0.03–0.19	**0.1**	0.04–0.24	**0.08**	0.04–0.18
White-chinned petrel	**0.28**	0.14–0.64	**0.08**	0.03–0.27	0.05	0.03–0.09	0.05	0.03–0.09	0.04	0.02–0.07	0.05	0.03–0.09
Yellow-eyed penguin	**0.08**	0.03–0.21	**0.09**	0.03–0.21	**0.08**	0.03–0.18	**0.09**	0.03–0.23	**0.11**	0.04–0.25	**0.08**	0.03–0.19
Northern royal albatross	**0.19**	0.07–0.59	**0.06**	0.01–0.20	**0.04**	0.01–0.15	**0.04**	0.01–0.16	0.01	0.00–0.06	**0.04**	0.01–0.16
Chatham petrel	**0.07**	0.00–0.32	**0**	0.00–0.11	0	0.00–0.00	0	0.00–0.00	0	0.00–0.00	0	0.00–0.00
Chatham Island taiko	**0**	0.00–0.36	0	0.00–0.00	0	0.00–0.00	0	0.00–0.00	0	0.00–0.00	0	0.00–0.00

Table 8.32: (from Richard et al. 2017). Potential Biological Removal (PBR_1, i.e., with a recovery factor f = 1), total annual potential fatalities (APF) in trawl, longline, and set-net fisheries, risk ratio with f = 1 (RR = APF/PST), and the probability that APF > PST with f = 1, f = 0:5 and f = 0:1 (P1, P0:5 and P0:1, respectively). Species are ordered in decreasing order of the median risk ratio. 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 was between 600 and 800. Species names are coloured according to their risk category. Red: risk ratio with a median over 1 or upper 95% confidence 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 are rounded to three significant digits

	PBR_ρ		APF		Risk ratio		P_1	$P_{0.5}$	$P_{0.1}$
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.			
Black petrel	437	220–834	468	316–666	1.15	0.51–2.03	0.63	0.98	1.00
Salvin's albatross	3 600	2 710–4 940	2 780	2 030–3 760	0.78	0.51–1.09	0.08	0.98	1.00
Flesh-footed shearwater	1 450	1 030–2 000	987	623–1 560	0.67	0.39–1.15	0.08	0.86	1.00
Westland petrel	350	234–520	180	67–407	0.48	0.18–1.19	0.06	0.46	1.00
Southern Buller's albatross	1 370	901–2 160	528	371–745	0.39	0.22–0.66	0.00	0.19	1.00
Chatham Island albatross	425	296–623	155	89–246	0.36	0.18–0.66	0.00	0.16	1.00
NZ white-capped albatross	10 900	7 630–15 800	3 830	2 690–5 380	0.35	0.21–0.58	0.00	0.08	1.00
Gibson's albatross	496	331–736	166	106–242	0.34	0.19–0.59	0.00	0.08	1.00
Northern Buller's albatross	1 630	1 050–2 570	397	294–523	0.25	0.14–0.41	0.00	0.00	1.00
Antipodean albatross	364	251–513	74	45–115	0.20	0.11–0.36	0.00	0.00	0.99
Yellow-eyed penguin (mainland)	121	80–179	23	8–47	0.18	0.07–0.45	0.00	0.01	0.89
Otago shag	285	182–425	41	22–64	0.14	0.07–0.28	0.00	0.00	0.85
Northern giant petrel	336	159–805	47	14–112	0.14	0.03–0.47	0.00	0.02	0.69
Spotted shag	3 710	1 780–6 900	335	215–484	0.09	0.04–0.20	0.00	0.00	0.43
Yellow-eyed penguin	287	191–425	23	8–47	0.08	0.03–0.19	0.00	0.00	0.29
Campbell black-browed albatross	1 980	1 010–3 590	153	88–264	0.08	0.04–0.18	0.00	0.00	0.27
White-chinned petrel	25 600	16 300–41 100	1 360	1 080–1 720	0.05	0.03–0.09	0.00	0.00	0.01
Northern royal albatross	716	342–1 360	34	10–92	0.04	0.01–0.16	0.00	0.00	0.09
Foveaux shag	207	130–316	8	2–15	0.04	0.01–0.08	0.00	0.00	0.01
Grey petrel	5 530	3 220–9 140	203	123–340	0.04	0.02–0.08	0.00	0.00	0.01
Southern royal albatross	848	596–1 170	19	6–41	0.02	0.01–0.05	0.00	0.00	0.00
Snares Cape petrel	1 600	602–4 030	19	2–74	0.01	0.00–0.06	0.00	0.00	0.01
Little black shag	338	153–655	3	0–11	0.01	0.00–0.04	0.00	0.00	0.00
Pied shag	1 120	702–1 680	9	0–29	0.01	0.00–0.03	0.00	0.00	0.00
Grey-faced petrel	29 900	19 200–49 500	146	57–321	0.00	0.00–0.01	0.00	0.00	0.00
Fluttering shearwater	36 100	15 100–72 700	140	68–272	0.00	0.00–0.01	0.00	0.00	0.00
Fiordland crested penguin	636	295–1 230	4	0–23	0.00	0.00–0.04	0.00	0.00	0.01
Sooty shearwater	617 000	291 000–1 240 000	1 470	790–2 810	0.00	0.00–0.01	0.00	0.00	0.00
Grey-headed albatross	695	349–1 250	3	0–15	0.00	0.00–0.03	0.00	0.00	0.00
Common diving petrel	135 000	47 800–309 000	317	46–1 250	0.00	0.00–0.01	0.00	0.00	0.00
Light-mantled sooty albatross	869	666–1 120	3	0–15	0.00	0.00–0.02	0.00	0.00	0.00
Hutton's shearwater	15 000	9 300–23 400	22	4–80	0.00	0.00–0.01	0.00	0.00	0.00
Northern little penguin	1 510	934–2 330	2	0–8	0.00	0.00–0.01	0.00	0.00	0.00
White-headed petrel	34 300	16 600–66 100	19	6–40	0.00	0.00–0.00	0.00	0.00	0.00
Southern little penguin	1 520	918–2 380	1	0–7	0.00	0.00–0.00	0.00	0.00	0.00
NZ white-faced storm petrel	332 000	137 000–669 000	131	28–376	0.00	0.00–0.00	0.00	0.00	0.00
Australasian gannet	9 440	4 240–19 300	5	0–20	0.00	0.00–0.00	0.00	0.00	0.00
Buller's shearwater	56 000	34 500–103 000	18	6–47	0.00	0.00–0.00	0.00	0.00	0.00
Southern black-backed gull	334 000	137 000–703 000	87	29–200	0.00	0.00–0.00	0.00	0.00	0.00
Fairy prion	329 000	214 000–506 000	127	15–566	0.00	0.00–0.00	0.00	0.00	0.00
Little shearwater	21 800	14 100–33 100	5	1–12	0.00	0.00–0.00	0.00	0.00	0.00
Black-bellied storm petrel	15 600	8 850–25 700	4	0–15	0.00	0.00–0.00	0.00	0.00	0.00
Cook's petrel	49 400	27 400–89 300	14	0–76	0.00	0.00–0.00	0.00	0.00	0.00
Broad-billed prion	69 100	45 700–105 000	11	1–35	0.00	0.00–0.00	0.00	0.00	0.00
Antarctic prion	154 000	77 200–289 000	22	4–60	0.00	0.00–0.00	0.00	0.00	0.00
Snares crested penguin	6 840	4 770–9 620	1	0–4	0.00	0.00–0.00	0.00	0.00	0.00
Mottled petrel	47 700	30 400–77 600	7	0–41	0.00	0.00–0.00	0.00	0.00	0.00
Auckland Island shag	473	198–952	0	0–1	0.00	0.00–0.00	0.00	0.00	0.00
Bounty Island shag	26	14–44	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Subantarctic skua	67	44–100	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Caspian tern	170	96–282	0	0–1	0.00	0.00–0.01	0.00	0.00	0.00
Chatham Island shag	76	46–116	0	0–3	0.00	0.00–0.04	0.00	0.00	0.00
Campbell Island shag	492	230–944	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Chatham Island little penguin	1 510	935–2 390	1	0–5	0.00	0.00–0.00	0.00	0.00	0.00
White-flippered little penguin	466	275–737	0	0–3	0.00	0.00–0.01	0.00	0.00	0.00
Eastern rockhopper penguin	11 100	6 800–17 500	1	0–3	0.00	0.00–0.00	0.00	0.00	0.00
Erect-crested penguin	17 800	12 600–24 600	1	0–3	0.00	0.00–0.00	0.00	0.00	0.00
White-bellied storm petrel	232	105–458	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
White tern	26	15–43	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
South Georgian diving petrel	10	5–18	0	0–1	0.00	0.00–0.07	0.00	0.00	0.02
NZ king shag	39	24–61	0	0–3	0.00	0.00–0.07	0.00	0.00	0.01
Kerm. storm petrel	11	4–25	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Masked booby	52	28–89	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
NZ storm petrel	51	6–192	0	0–1	0.00	0.00–0.04	0.00	0.00	0.01
Pitt Island shag	104	61–161	0	0–2	0.00	0.00–0.03	0.00	0.00	0.00
Chatham petrel	42	23–76	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Chatham Island taiko	2	1–4	0	0–0	0.00	0.00–0.00	0.00	0.00	0.01
Pycroft's petrel	412	246–723	0	0–1	0.00	0.00–0.00	0.00	0.00	0.00
Soft-plumaged petrel	499	137–1 280	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Wedge-tailed shearwater	5 930	3 120–10 500	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00
Kerm. petrel	781	511–1 320	0	0–1	0.00	0.00–0.00	0.00	0.00	0.00
White-naped petrel	7 010	3 320–13 800	0	0–0	0.00	0.00–0.00	0.00	0.00	0.00

Table 8.33: (from Richard et al. 2017). Estimated annual potential seabird fatalities by fishing method and target species/species group.

Fishing method	Target	Total annual potential fatalities	95% c.i.
Trawl	Inshore	4 800	3 140–7 080
	Hoki	1 540	1 140–2 050
	Flatfish	1 210	804–1 820
	Middle depth	1 060	777–1 410
	Scampi	777	489–1 150
	Squid	775	561–1 020
	Deepwater	237	120–433
	Ling	174	121–252
	Hake	91	62–124
	Southern blue whiting	80	41–140
	Jack mackerel	38	19–65
BLL	Small vessel, ling	1 090	809–1 410
	Snapper	523	385–681
	Minor targets	254	170–364
	Hapuka	238	140–398
	Large vessel, ling	192	130–268
	Bluenose	125	49–223
SLL	Bigeye	422	318–548
	Small vessel, southern bluefin	359	277–450
	Swordfish	262	170–372
	Large vessel, southern bluefin	33	19–49
	Minor targets	9	4–16
	Albacore	2	0–5
SN	Shark	42	24–65
	Flatfish	31	13–62
	Minor targets	21	9–39
	Grey mullet	4	0–11
Total	Total	14 400	11 900–17 500

Table 8.34: (from Richard et al. 2017). Estimated number of annual observable captures of seabirds (not including cryptic mortality), and estimated number of annual potential fatalities (including cryptic mortality) in trawl, bottom-longline, surface-longline and set-net fisheries in New Zealand's Exclusive Economic Zone. The species names are coloured according to their respective risk categories: Red: risk ratio with a median over 1 or upper 95% confidence 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.

Species	No cryptic mortality		With cryptic mortality	
	Mean	95% c.i.	Mean	95% c.i.
Gibson's albatross	93	64–125	166	106–242
Antipodean albatross	40	25–57	74	45–115
Southern royal albatross	7	1–15	19	6–41
Northern royal albatross	11	3–26	34	10–92
Campbell black-browed albatross	54	31–87	153	88–264
NZ white-capped albatross	432	358–519	3 830	2 690–5 380
Salvin's albatross	434	351–529	2 780	2 030–3 760
Chatham Island albatross	56	28–96	155	89–246
Grey-headed albatross	1	0–5	3	0–15
Southern Buller's albatross	124	96–154	528	371–745
Northern Buller's albatross	144	107–191	397	294–523
Light-mantled sooty albatross	1	0–4	3	0–15
Northern giant petrel	17	4–45	47	14–112
Grey petrel	79	49–119	203	123–340
Black petrel	248	189–318	468	316–666
Westland petrel	52	27–88	180	67–407
White-chinned petrel	697	598–803	1 360	1 080–1 720
Flesh-footed shearwater	338	259–441	987	623–1 560
Wedge-tailed shearwater	0	0–0	0	0–0
Buller's shearwater	7	2–14	18	6–47
Sooty shearwater	355	301–420	1 470	790–2 810
Fluttering shearwater	68	36–107	140	68–272
Hutton's shearwater	6	1–14	22	4–80
Little shearwater	2	0–6	5	1–12
Snares Cape petrel	2	0–8	19	2–74
Fairy prion	17	2–63	127	15–566
Antarctic prion	5	1–10	22	4–60
Broad-billed prion	4	0–14	11	1–35
Pycroft's petrel	0	0–0	0	0–1
Cook's petrel	2	0–10	14	0–76
Chatham petrel	0	0–0	0	0–0
Mottled petrel	1	0–4	7	0–41
White-naped petrel	0	0–0	0	0–0
Kerm. petrel	0	0–1	0	0–1
Grey-faced petrel	43	20–74	146	57–321
Chatham Island taiko	0	0–0	0	0–0
White-headed petrel	6	1–14	19	6–40
Soft-plumaged petrel	0	0–0	0	0–0
Common diving petrel	20	6–50	317	46–1 250
South Georgian diving petrel	0	0–0	0	0–1
NZ white-faced storm petrel	37	4–137	131	28–376
White-bellied storm petrel	0	0–0	0	0–0
Black-bellied storm petrel	1	0–3	4	0–15
Kerm. storm petrel	0	0–0	0	0–0
NZ storm petrel	0	0–0	0	0–1
Yellow-eyed penguin	22	8–42	23	8–47
Northern little penguin	1	0–6	2	0–8
White-flippered little penguin	0	0–2	0	0–3
Southern little penguin	1	0–6	1	0–7
Chatham Island little penguin	0	0–3	1	0–5
Eastern rockhopper penguin	0	0–2	1	0–3
Fiordland crested penguin	3	0–14	4	0–23
Snares crested penguin	1	0–4	1	0–4
Erect-crested penguin	0	0–2	1	0–3
Australasian gannet	4	0–15	5	0–20
Masked booby	0	0–0	0	0–0
Pied shag	7	0–24	9	0–29
Little black shag	3	0–10	3	0–11
NZ king shag	0	0–2	0	0–3
Otago shag	32	18–49	41	22–64
Foveaux shag	6	1–12	8	2–15
Chatham Island shag	0	0–2	0	0–3
Bounty Island shag	0	0–0	0	0–0
Auckland Island shag	0	0–1	0	0–1
Campbell Island shag	0	0–0	0	0–0
Spotted shag	260	178–355	335	215–484
Pitt Island shag	0	0–2	0	0–2
Subantarctic skua	0	0–0	0	0–0
Southern black-backed gull	39	15–71	87	29–200
Caspian tern	0	0–0	0	0–1
White tern	0	0–0	0	0–0

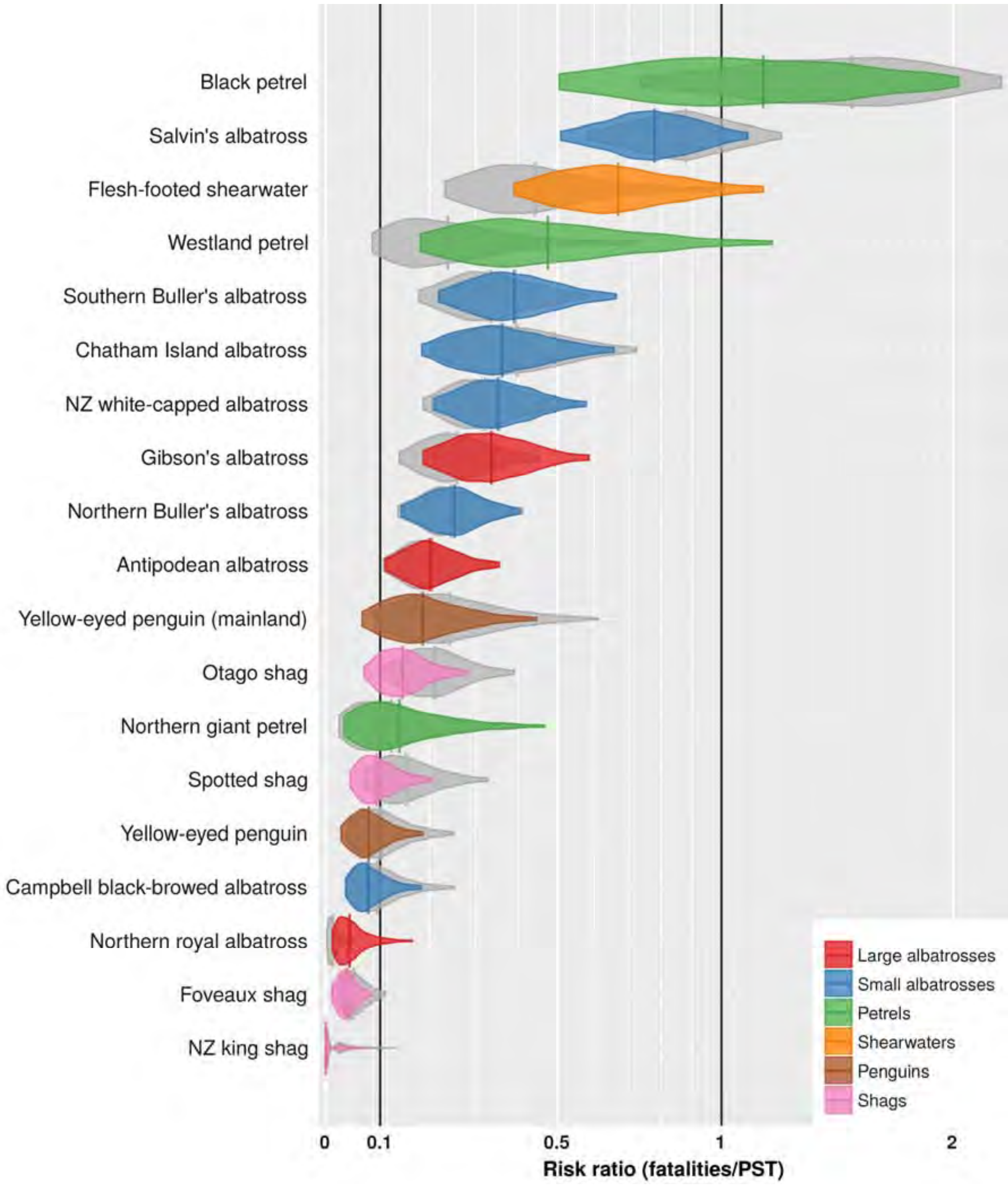


Figure 8.20: (from Richard et al. 2017). Risk ratio, showing the difference in risk resulting from the fishing data before and after the NPOA-seabirds while using the latest methodology, based on the base case model for the 2016 iteration of the risk assessment. The risk ratio is displayed on a logarithmic scale, with the threshold of the number of annual potential bird fatalities equalling the PST with *f* = 0:1 and *f* = 1 indicated by the two vertical black lines, and the distribution of the risk ratios within their 95% confidence interval indicated by the coloured shapes, including the median risk ratio (vertical line). Seabird species are listed in decreasing order of the median risk ratio. Species with a risk ratio of almost zero were not included (95% upper limit with *f* = 1 less than 0.1). 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 was between 600 and 800. The grey shapes indicate the risk ratios from the 2016 iteration base case as applied to fishing data from 2006–07 to 2010–11, using the base case model for the 2016 iteration of the risk assessment.

It should be noted that the Level 2 risk assessments conducted thus far (Richard et al. 2011, Richard & Abraham 2013b, 2015) include APFs in commercial fisheries within New Zealand's EEZ but exclude non-commercial impacts, fatalities on the High Seas and in other jurisdictions, and all other anthropogenic sources of mortality. Because of this focus and the definition of PBR as a level of mortality that can support all anthropogenic sources of mortality and still lead to good population outcomes, the risk ratios estimated by Richard & Abraham (2015) will be underestimates of the total risk faced by each taxon and interpretation should be in this context. Many of the other anthropogenic sources of mortality excluded from the risk assessment are poorly understood, although MPI will shortly commission a 'global' seabird risk assessment to include at least the commercial fishing components under project PRO2013-13.

8.4.5.2 FULLY QUANTITATIVE MODELLING

Fully quantitative population modelling has been conducted only for southern Buller's albatross, black petrel, white-capped albatross, and Gibson's (wandering) albatross. Data of similar quality and quantity are available for Antipodean (wandering) albatross, and this modelling will be conducted in 2015 under MPI project PRO2012-10, but 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.

8.4.5.2.1 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 (*Thalassarche bulleri bulleri*). They estimated (see also Sagar & Stahl 2005) that the adult population had increased about five-fold since about 1950 (Figure 8.21) 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 also in five colonies where counts were made in many non-census years.

The modelling was repeated in 2015 including eight years of additional mark-recapture data and a new census estimate in 2014. 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, 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.

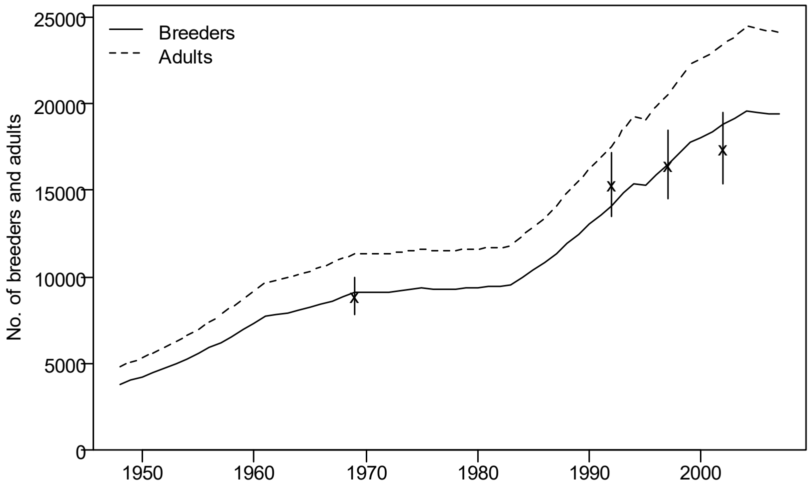


Figure 8.21: (from Francis et al. 2008). Estimates from model SBA21 of numbers of breeders (solid line) and adults (broken line) of southern Buller's albatross in each year. Also shown are the census observations (after Sagar & Stahl 2005) of numbers of breeders (crosses), with assumed 95% confidence intervals (vertical lines).

8.4.5.2.2 QUANTITATIVE MODELS FOR BLACK PETREL

Francis & Bell (2010) analysed data from the main population of black petrel (*Procellaria parkinsoni*), which breeds on 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 8.22). 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.

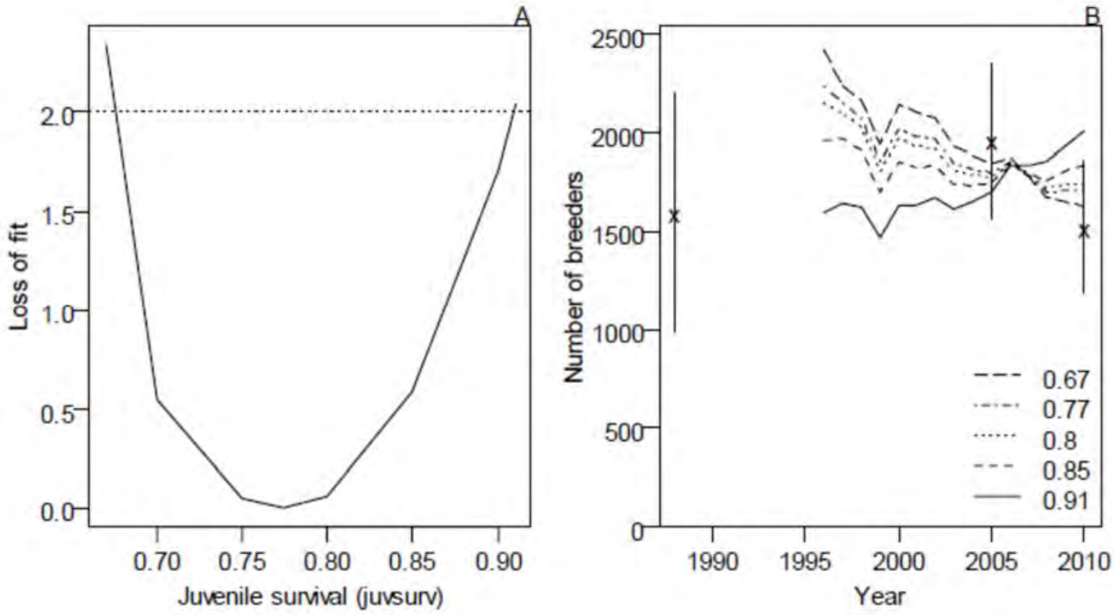


Figure 8.22: (from Bell et al. 2012). Likelihood profile for annual probability of juvenile survival of black petrel, showing: A, the loss of fit (the horizontal dotted line shows a 95% confidence interval for this parameter); and B, population trajectories corresponding to different values of juvenile survival, together with population estimates from transect counts (crosses with vertical lines indicating 95% confidence intervals. Note that the 1988 population estimate was not used in the model.

8.4.5.2.3 QUANTITATIVE MODELS FOR WHITE-CAPPED ALBATROSS

Francis (2012) described quantitative models for white-capped albatross (*Thalassarche steadi*), 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) suggest 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 (Baker et al. 2014b, Figure 8.23), which show considerable annual variation. Baker et al. (2014b) consider that the substantial year to year variation in counts is real, and that trend analyses appropriate in this situation support the null hypothesis of no trend in the population.

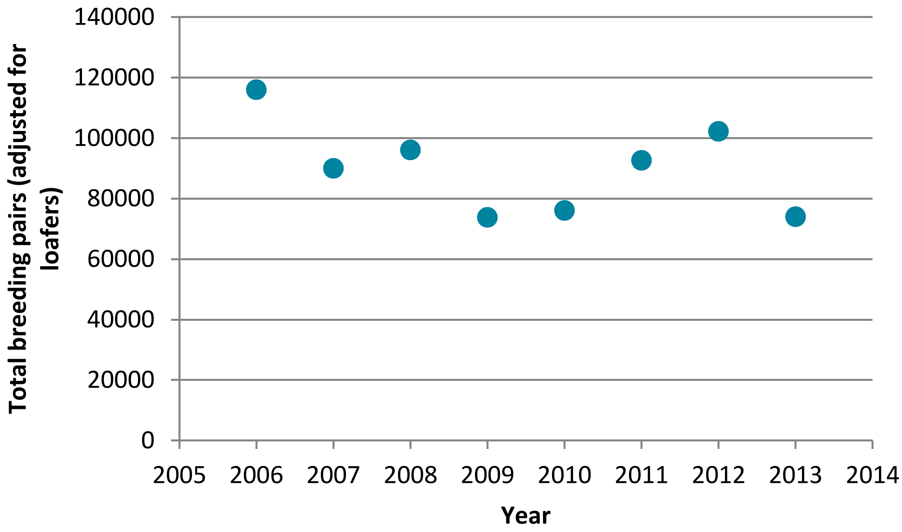


Figure 8.23: (Data from Baker et al. 2014b). Total counts of white-capped albatross at the Auckland Islands (as adjusted for the presence of non-breeding birds).

8.4.5.2.4 QUANTITATIVE MODEL FOR GIBSON'S ALBATROSS

Francis et al. (2015) concluded that there is cause for concern about the status of the population of Gibson's wandering albatross (*Diomedea gibsoni*) on the Auckland Islands. Since 2005, the adult population has been declining at 5.7%/yr (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 8.24). Forward projections showed that the most important of these to the future status of this population is adult survival (Figure 8.25).

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.

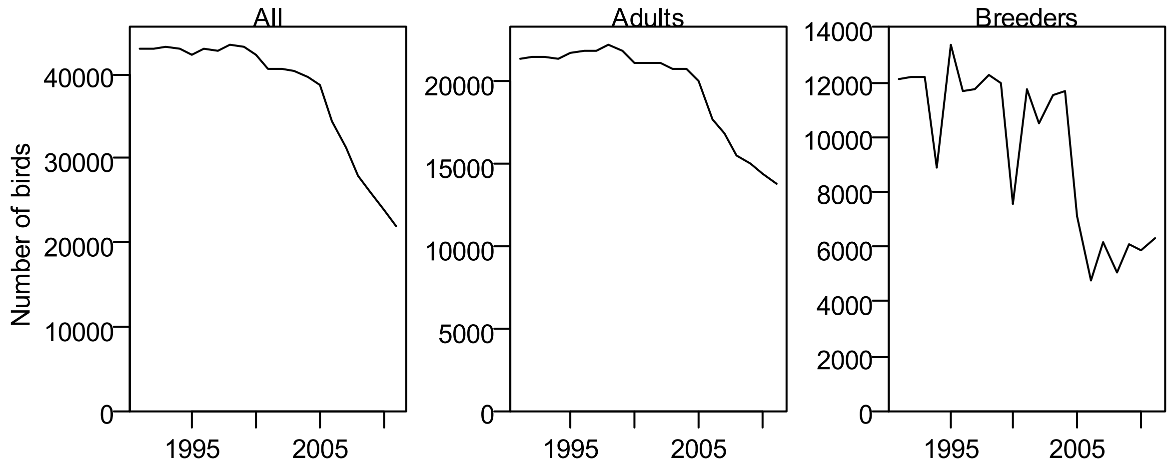


Figure 8.24: Estimated population trajectories for the whole Auckland Islands population of Gibson's wandering albatross. These were calculated by scaling up Francis et al.'s (2015) GIB5 trajectories to match the Walker & Elliott (1999) estimate for the whole population.

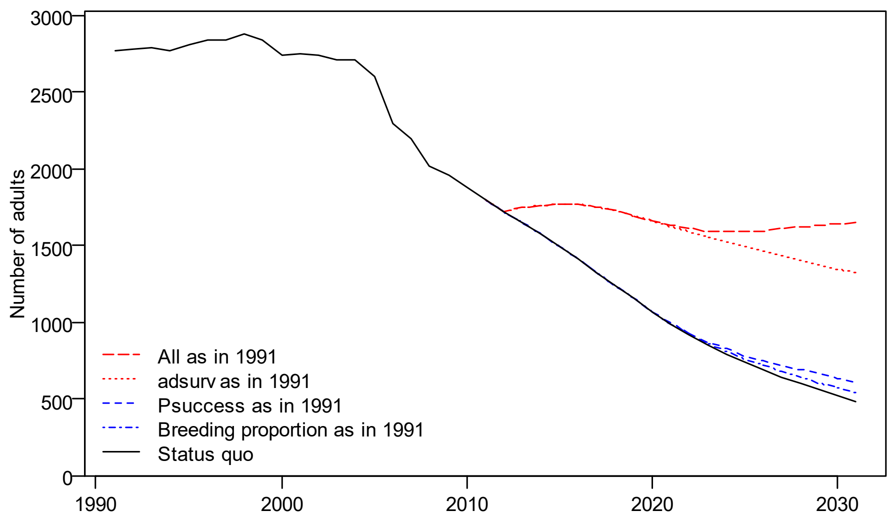


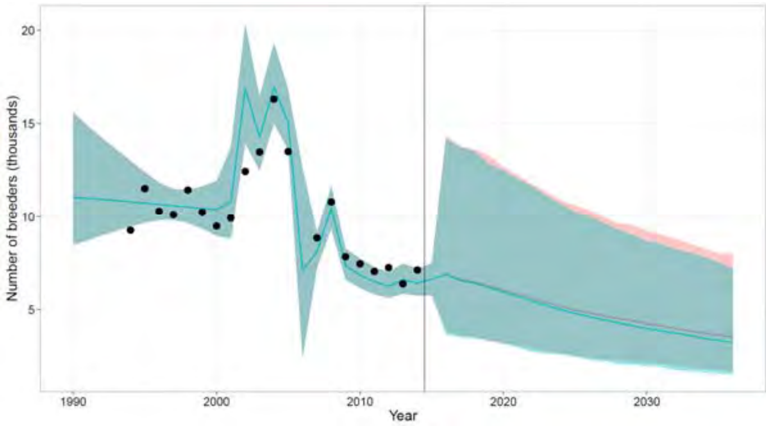
Figure 8.25: Estimated population trajectory for Gibson's albatross adults from Francis et al.'s (2015) model GIB5 with 20-year projections under five alternative scenarios about three demographic parameters: adult survival (adsurv); breeding success (Psuccess); and proportion of adults breeding. These scenarios differ according to whether each parameter remains at its status quo (i.e., 2011) level or recovers immediately to its 1991 level.

8.4.5.2.5 QUANTITATIVE MODEL FOR ANTIPODEAN ALBATROSS

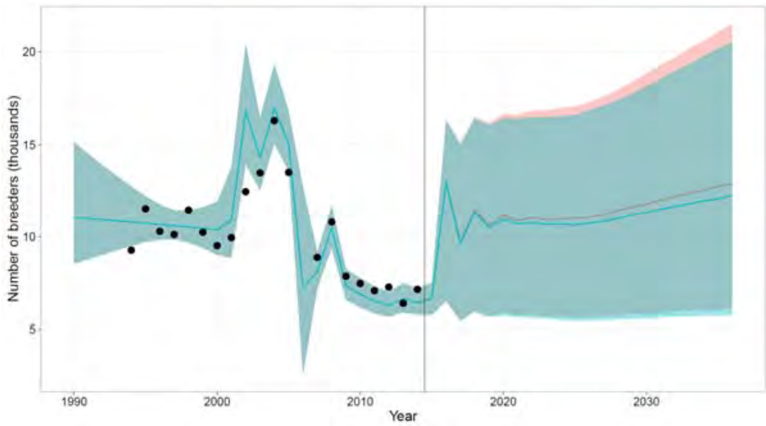
Edwards et al. (2017) developed a quantitative demographic model for the Antipodean wandering albatross (*D. antipodensis antipodensis*) to estimate vital rates and predict population changes into the future given the observed declines in the population since 2005 (Elliott & Walker 2014). The model was 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, 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 8.26). 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 of New Zealand waters could not be evaluated due to a lack of data (Edwards et al. 2017).

(a) Basecase model with vital rates sampled from 2004–13



(b) Basecase model with vital rates from 1995–2004



(c) Female-only model with vital rates sampled from 2004–13

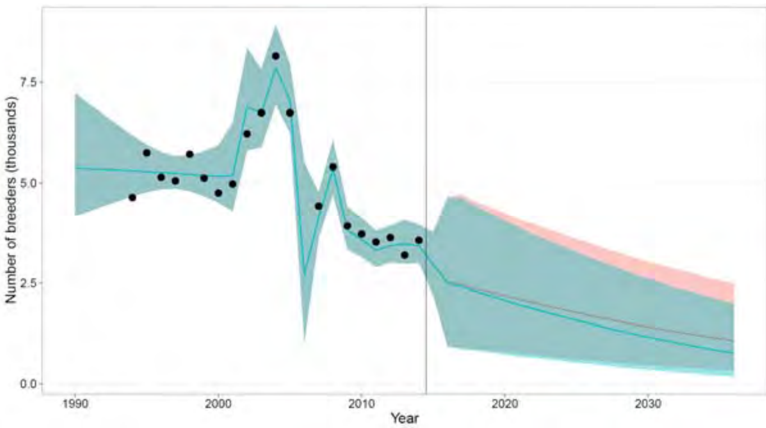


Figure 8.26: Predicted population dynamics for the whole island, showing the number of Antipodean albatross breeders from 2015–34. Predicted dynamics with and without fishing pressure are superimposed. The median and 95% confidence intervals are shown for each scenario. The vertical line indicates the limit of the empirical data, with subsequent dynamics representing an extrapolation. Empirical census data are also shown.

8.4.5.2.6 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.

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.

Fletcher et al. (2008) sought to assess the potential impact of fisheries on Antipodean and Gibson's wandering albatrosses (*Diomedea antipodensis antipodensis* and *D. a. gibsoni*); black petrel (*Procellaria parkinsoni*) 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 0.3k 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 PBR 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 (Table 8.34). 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.

8.4.5.2.7 GENERAL CONCLUSIONS FROM QUANTITATIVE MODELLING

Fully quantitative modelling has now been conducted for four of the five seabird populations for which apparently suitable data are available. 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) and another is declining quite rapidly (Gibson's 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, can be summarised as:

- Very few seabird populations have sufficient data for fully quantitative modelling.
- Except for the two most complete datasets (southern Buller's and Gibson's albatross) it has been difficult to draw firm conclusions about trends in population size.
- 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.

8.4.5.3 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.

8.4.5.3.1 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 & Abraham (2015) provided plots of such an exploration for nine taxa (Figure 8.27). 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.

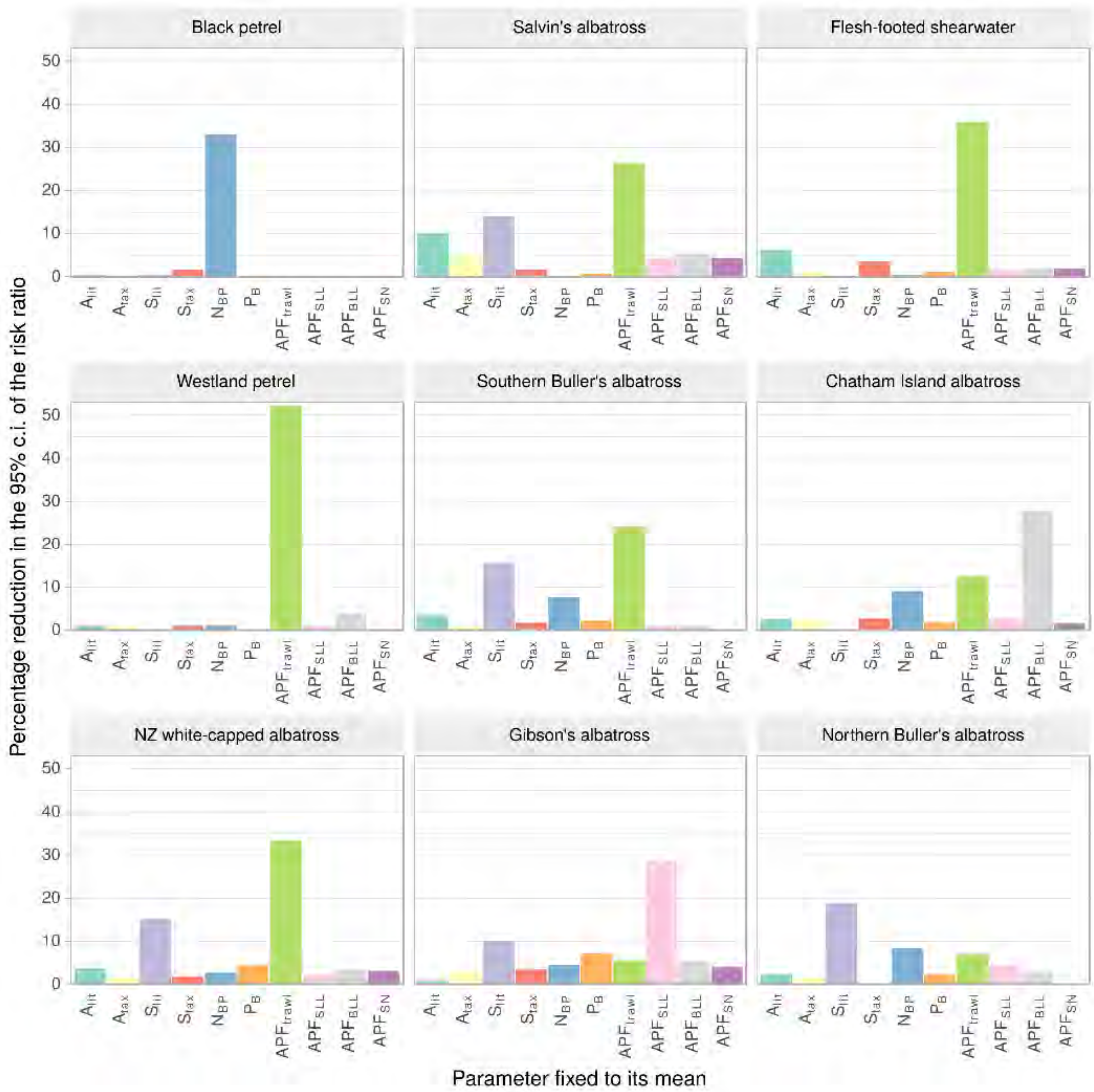


Figure 8.27: (reproduced from Richard et al. 2017). Sensitivity of the uncertainty in the risk ratio for the nine seabird species with the highest risk ratio. For each seabird type, the sensitivity to the uncertainty in the following parameters is considered: annual potential fatalities in trawl, bottom-longline, 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.

8.4.5.3.2 SCARCITY OF INFORMATION ON CRYPTIC MORTALITY

Cryptic mortality 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 fatal consequences).

Given the relatively small sample sizes in both of these trials, there is substantial (estimatable) uncertainty in the estimates from the trials themselves and additional (non-estimatable) 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 Brothers et al.'s (2010) longline trial is 1.8–2.5 (mean 2.1), and that for Watkins et al.'s 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 every one observed captured, and 6440 (3400–20 000) warp strikes by small birds for every 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 risk assessment (Richard & Abraham 2013b).

A review of available information on cryptic mortality has been commissioned under CSP project INT2013-05 and supported by MPI project PRO2012-17. The final report was not available to be included at the time of this review.

8.4.5.3.3 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 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 8.28).

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 Abraham et al.'s (2010a) interviews were conducted, there were an estimated 4.8 (4.4–5.2) million fisher hours rod and line fishing from trailer boats in 2004–05 (Hartill et al. 2007). Applying Abraham et al.'s (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.

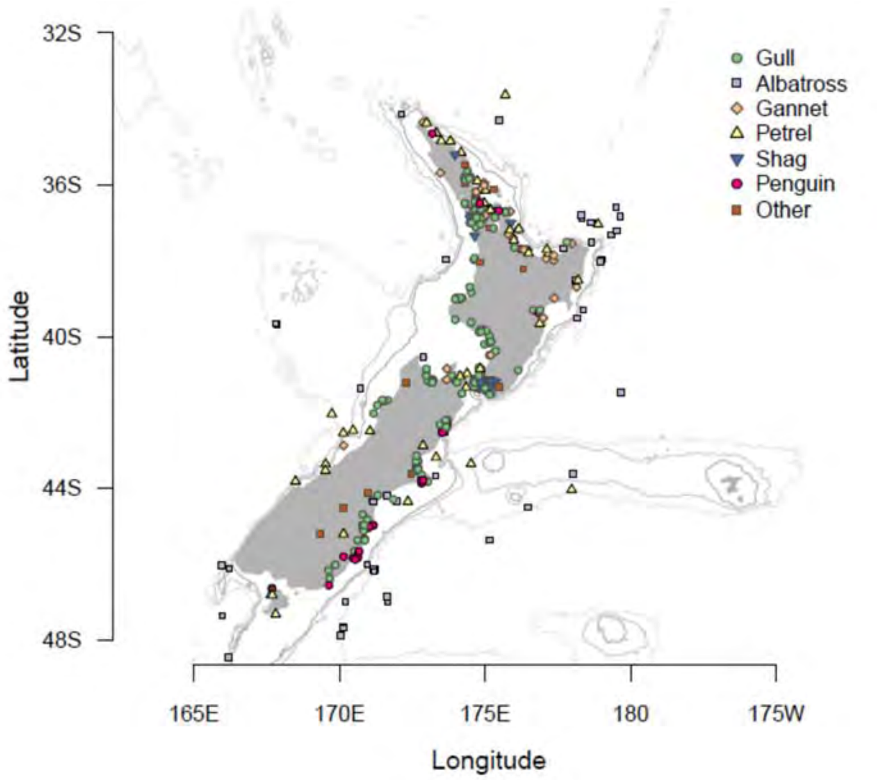


Figure 8.28: (from Abraham et al. 2010a). Distribution of the reported capture locations for banded seabirds reported as being captured in fishing gear, 1952–2007. Note, band recovery locations are reported with low spatial precision and some of the inland locations may be correct.

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 two 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.

8.4.5.3.4 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: 4 used the South Atlantic; 4 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 8.35), and more than 10 000 white-chinned petrel are killed off South America each year (Phillips et al. 2006), noting that reliable records are not available for most of the fisheries involved. Also note that white-chinned petrels also breed on Prince Edward and Falkland Islands, South Georgia, Iles Crozet, and the Kerguelen group, so South American captures may be from other 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).

Table 8.35: (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

8.4.5.3.5 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 in DOC's Action Plan for Seabird Conservation in New Zealand (Taylor 2000) and various Threatened Species Recovery Plans.

8.4.5.3.6 FUTURE DEVELOPMENT OF THE RISK ASSESSMENT FRAMEWORK

The following steps were identified in the NPOA-seabirds 2013 (MPI 2013) in order to improve the risk assessment framework that supports the implementation of the NPOA-seabirds 2013:

- implementation of a framework and process to consolidate different risk assessment and population monitoring results into an integrated assessment, including:
  - checking the algorithmic Level 2 assessment results for 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.

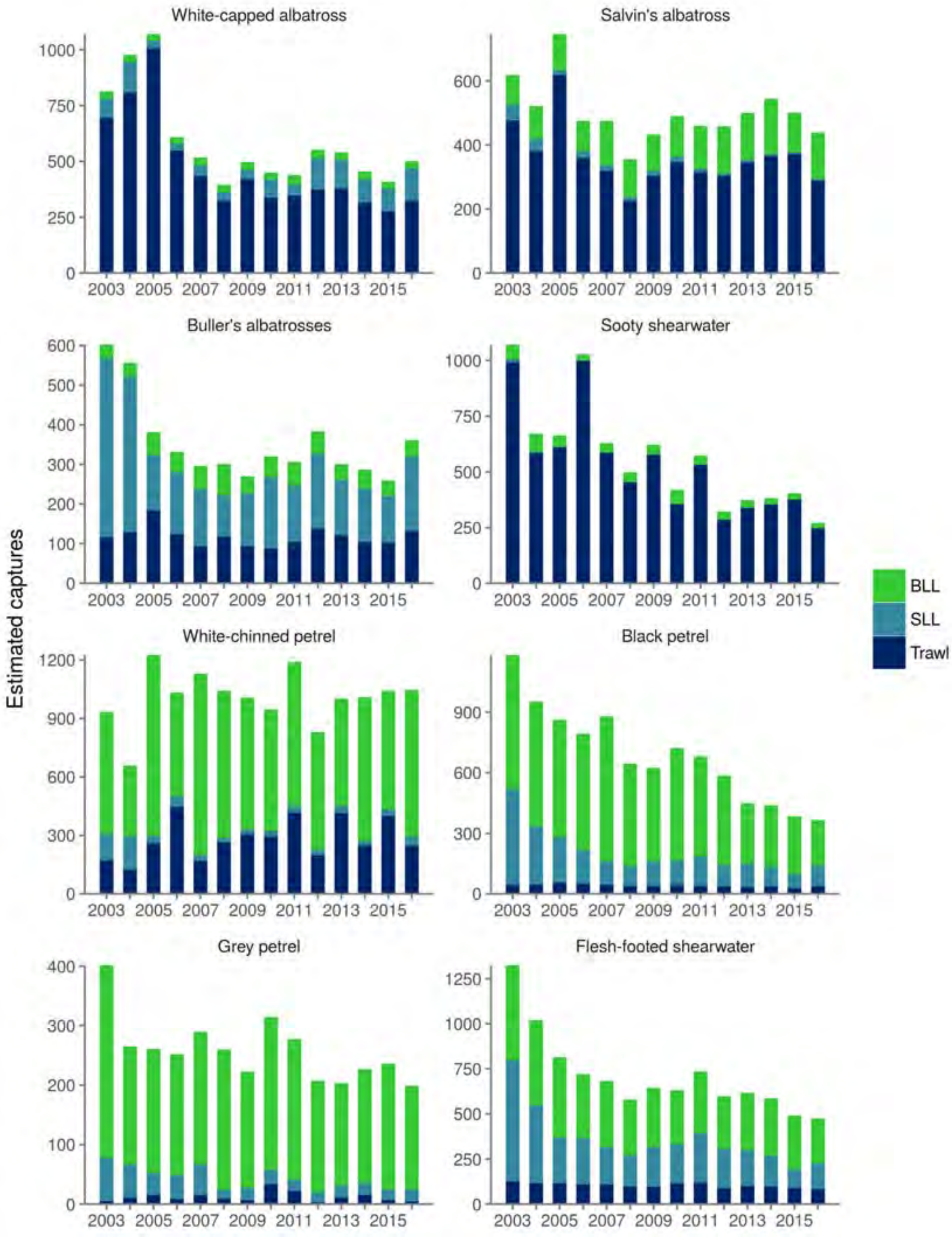
MPI has committed to undertake further iterations of the seabird Level 2 risk assessment in 2017. MPI intends to arrange an international review of the risk assessment.

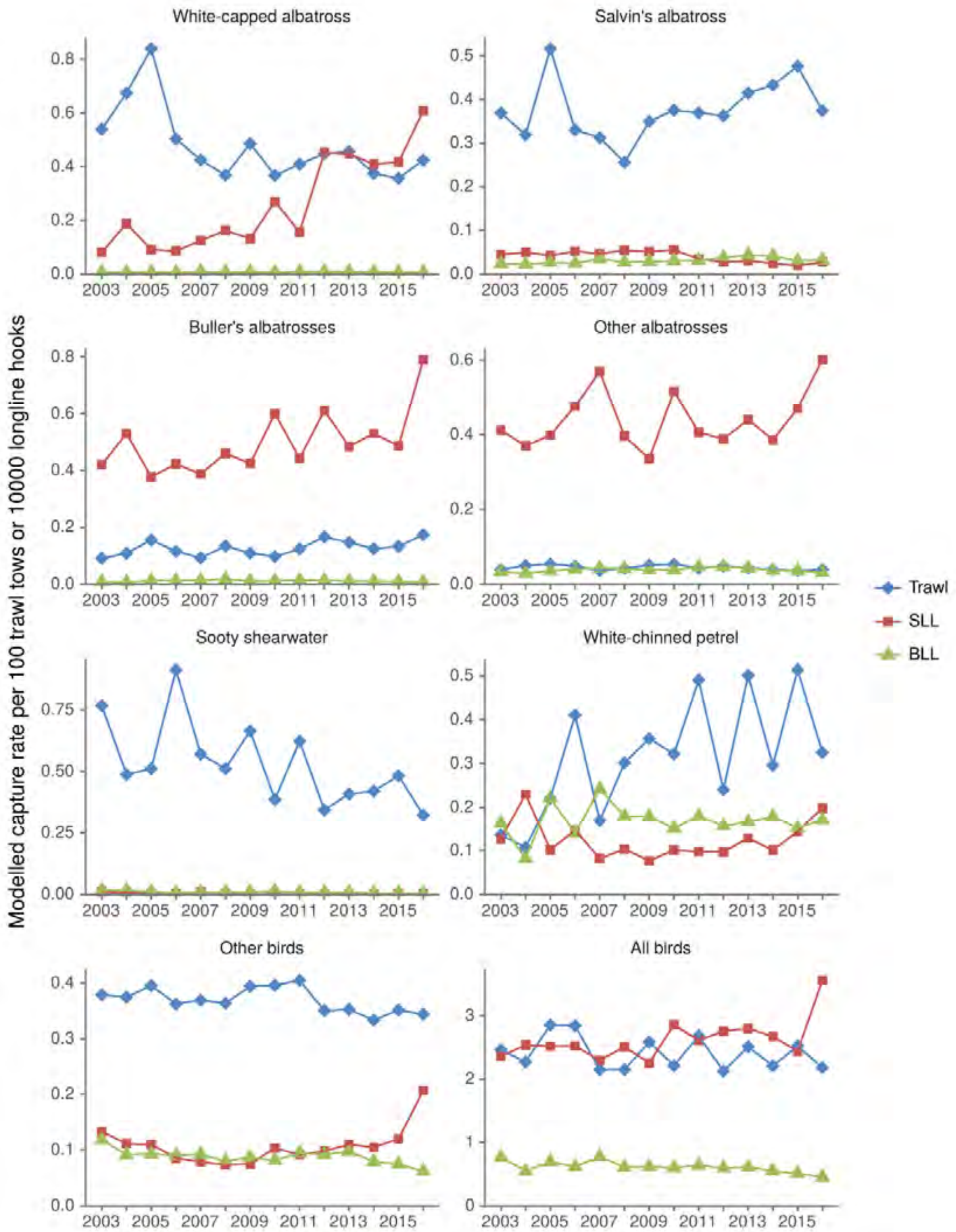
8.5 INDICATORS AND TRENDS

Population size	Multiple species and populations: see Taylor (2000)
Population trend	Multiple species and populations: see Taylor (2000)
Threat status	Multiple species and populations: see Robertson et al. (2017)
Number of interactions[1]	In the 2014–15 October fishing year, there were an estimated 4378 seabird captures (excluding cryptic mortalities) across all trawl and longline fisheries (http://data.dragonfly.co.nz/psc, data version v2016001). About 45% of the estimated captures across these fisheries (other fisheries such as set net are excluded) were in trawl fisheries, 13% in surface-longline fisheries, and 42% in bottom-longline fisheries:

Bird group	Trawl	Surface longline	Bottom longline	All these methods
White-capped albatross	439	264	345	1 048
Salvin's albatross	530	308	407	1 245
Buller's albatross	167	75	112	354
Other albatrosses	99	109	155	363
Sooty shearwater	463	292	364	1 119
White-chinned petrel	185	25	444	654
Black petrel	47	52	293	392
Grey petrel	16	24	168	208
Flesh footed shearwater	116	181	340	637
Other birds	587	18	418	1 023
All birds combined	2 277	660	2 138	5 075

[1] For more information, see: http://data.dragonfly.co.nz/psc.

Trends in interactions	Captures of all birds combined show a decreasing trend between 2002–03 and 2014–15 (http://data.dragonfly.co.nz/psc, data version v2016001) but there are substantial differences in trends between species and fisheries. Captures of white-capped albatross and sooty shearwater have decreased, especially in offshore trawl fisheries: 

Trends in interactions [Continued]	Capture rate trends (excluding cryptic mortalities) are described for the four fisheries estimated to account for most captures of a species (usually accounting for 70–80% of the total). Capture rates of white-capped albatross have fallen in trawl fisheries for hoki and squid but have remained steady in inshore trawl fisheries and increased in the southern bluefin tuna longline fishery. Capture rates for other albatross species for which specific estimates were made (Salvin's and Buller's) have fluctuated without obvious trend in trawl and bottom-longline fisheries but increased in surface-longline fisheries. Capture rates for white-chinned petrel have increased in the squid trawl fishery but have remained steady in longline fisheries. Capture rates of sooty shearwater have declined in the ling longline fishery but have fluctuated without apparent trend in other key fisheries. 

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THEME 2: NON-PROTECTED BYCATCH

9 FISH AND INVERTEBRATE BYCATCH

<table>
<tr><td>Scope of chapter</td><td>This chapter outlines the main non-protected species (fish and invertebrates) caught as bycatch in New Zealand’s major offshore fisheries, with summaries of the amounts caught and discarded. Small amounts of some protected species (e.g., deep-sea corals) may also be included. Much of the research in this field has been conducted fishery by fishery with no spatial breakdown of annual catch totals, but this summary incorporates a re-assessment of historical analyses for deepwater fisheries, with stratification aligned to standardised areas. This provides estimates of bycatch and discarding across all offshore fisheries within separate regions of New Zealand fisheries. Research begun in 2013 to analyse individual species bycatch over time for each of the Tier 1 Deepwater fisheries has continued, with updated estimates for some fisheries and for the 2013–14 fishing year. This chapter presents the latest available information, however the last date of detailed analysis differs between fisheries, e.g., the Hoki/Hake/Ling fishery was updated to 2012–13, the Jack Mackerel fishery to 2013–14 (see Table 9.1 for more details).

The fisheries summarised are as follows:
<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>Area</td><td>Total annual bycatch and discard levels are summarised for 10 of the 11 fishery areas shown in Figure 9.1. Bycatch from the small amount of fishing in the Kermadec area is not addressed.

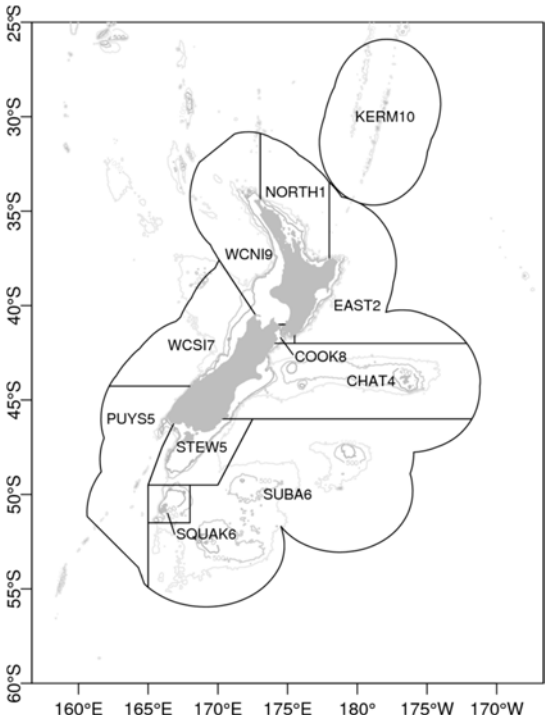

Figure 9.1: Standardised assessment areas for estimation of total non-protected fish and invertebrate bycatch in offshore fisheries.</td></tr>
</table>

Focal localities	**Trawl fisheries** *Arrow squid*: Auckland Islands and Stewart/Snares Shelf (80–300 m). *Hoki/hake/ling*: Chatham Rise, west coast South Island, Campbell Plateau, Puysegur Bank, and Cook Strait (200–800 m). *Jack mackerel*: West coast of the North and South Islands, 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 and South Islands, Chatham Rise, and Auckland Islands (300–450 m). **Longline fisheries** *Ling (bottom)*: Chatham Rise, Bounty Plateau, and Campbell Plateau (150–600 m). *Tuna (surface)*: East coast of the North Island and west coast of the South Island. **Other fisheries** *Albacore tuna troll*: West coasts of the North and South Islands. *Skipjack tuna purse seine*: Northern North Island
Key issues	• Lack of data on bycatch 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. Work is underway to implement electronic monitoring tools on inshore vessels for data collection which may help address this issue.
Emerging issues	• Trends of increased rates and levels of bycatch and discarding in several categories of catch, especially non-QMS fish species and invertebrates. • The effect on bycatch rates in the ling longline fishery of a change to heavier fishing gear (including integrated weights) as used in the Antarctic toothfish fishery.
MPI research (current)	HMS2013-01 *Bycatch in tuna longline fisheries*; ENV2015-04 *Estimation of incidental captures, fish bycatch and discards using electronic monitoring*; DAE201601 *Total catch composition in depwater fisheries (arrow squid & scampi)*; BEN201401 *Benthic risk assessment*; ENV201501 *Updating tools for fish identification*;DAE2017-01 *Bycatch monitoring and quantification in the HOK/HAK/LIN* (upcoming); DAE2017-02 *Taxonomic identification of benthic samples* (upcoming); DAE2017-04 *Quantification of key bycatch groups across fisheries* (upcoming).
NZ government research (current)	DOC18303 *Age & growth of NZ protected corals*; DOC16307 *Identification of coldwater corals*; SEA201622 *Habitat suitability mapping of coldwater corals - high seas*.
Related chapters/issues	Chondrichthyans (sharks, rays, and chimaeras)

Note: This chapter has been updated for the AEBAR 2017

1.1 CONTEXT

For this chapter *non-target fish species catch* is equivalent to *bycatch*, defined as ‘all fish caught that were not the stated target species for that tow 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). *Discarded catch* in this report includes estimates of any fish lost from the net at the surface. Estimates of *non-target catch*, if required, can be obtained from this report by adding target species *discards* to total *bycatch*.

DEEPWATER TRAWL AND BOTTOM-LONGLINE FISHERIES

The management of non-protected species bycatch in the deepwater and middle-depth fisheries is described in the National Fisheries Plan for Deepwater and Middle-depth Fisheries (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of non-protected species bycatch is Management Objective 2.4: *Identify and avoid or minimise adverse effects of deepwater and middle-depth fisheries on incidental bycatch species*. Specific objectives for the management of non-protected species bycatch are outlined in the fishery-specific chapters of the National Deepwater Plan. Estimation of non-protected species bycatch 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
- 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-protected fish species bycatch in the fisheries for Highly Migratory Species (HMS) is addressed in the HMS fish plan. Tuna fisheries incidental bycatch is examined, with updates every 2–3 years planned. Some data on bycatch in the Albacore tuna troll fishery and the skipjack tuna purse seine fishery are also available.

INSHORE FISHERIES

The three National Fisheries Plans for Inshore species (finfish, shellfish and freshwater fisheries) also include objectives that address non-protected species bycatch, but research on these objectives has 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 elephantfish (Starr et al. 2010a, 2010b, 2010c, Starr & Kendrick 2012, Starr & Kendrick 2013).

1.2 GLOBAL UNDERSTANDING

Bycatch of unwanted zero- or low-value species and discarding of these and of target species that are damaged or too small to process are significant issues in many fisheries worldwide. Few, if any, fisheries are completely without bycatch 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 was 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.

Tropical shrimp trawl fisheries typically have 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. Discard rates in demersal trawl fisheries targeting finfish are much lower but, because they are so widespread, make a considerable contribution to total global discards. Tuna longline fisheries have the next largest contribution and tend to have greater unwanted bycatch than other line fisheries (Kelleher 2005).

The estimated global level of discards reduced considerably since the Alverson et al. (1994) estimate, but differences in the methodology and definition of bycatch used (see Kelleher 2005, Davies et al. 2009) make it difficult to quantify the decline. The main reasons for the estimated decline in bycatch may be due to a combination of higher retention rates, better fisheries management, and more selective/targeted fishing methods.

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 applied in some New Zealand fisheries are frequently used to scale observed bycatch and discard rates to the wider fishery, and the methods used in New Zealand fisheries are 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 assessment (Anderson & Edwards, in prep).

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 STATE OF KNOWLEDGE IN NEW ZEALAND

1.3.1 OVERVIEW

Estimation of annual bycatch and discard levels of non-protected species in selected New Zealand fisheries have been undertaken at regular intervals since 1998 (Table 9.1).

Table 9.1: Summary of research into bycatch and discards in New Zealand fisheries.

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 (in prep)
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)
Hake trawl (HAK)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017)
Ling trawl (LIN)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017)
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)
Jack mackerel trawl (JMA)	Anderson et al. (2000) Anderson (2004b) Anderson (2007) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson et al. (2017b) Anderson (2017)
Southern blue whiting trawl (SBW)	Clark et al. (2000) Anderson (2004a) Anderson (2009b) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017)
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)
Oreo trawl (OEO)	Clark et al. (2000) Anderson (2004a) Anderson (2011) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Anderson et al. (2017a)
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 (in prep)

Other fisheries	Report
Albacore tuna troll	Griggs et al. (2014)
Skipjack tuna purse seine	Anon (2013)
Skipjack tuna purse seine	Anon (2017)

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 fishery strata in the observed fraction of the fishery, and 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, while 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, but without estimates of precision. The original analyses were based on a stratification using different sets of areas and in some cases 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 has 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 of a wide range of individual species were also made in the most recent analysis of the arrow squid trawl fishery (Anderson 2013a), ling longline fishery (Anderson 2014a), hoki/hake/ling fishery (Ballara 2015), jack mackerel fishery (Anderson et al. 2017b), orange roughy and oreo fisheries (Anderson et al. 2017a), and the scampi and arrow squid fisheries (Anderson & Edwards, in prep) as well as in a more simplified manner for the remaining Deepwater Tier 1 fisheries (Anderson 2017).

In some cases the apparent increase or decrease in bycatch of a species is likely 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 seemingly 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 as 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 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. In this case CPUE was calculated as the number of fish observed caught per 1000 hooks set stratified by fishing year, fleet (Foreign Licenced, 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 deepwater trawl and bottom-longline fisheries are currently not available.

TROLL AND PURSE SEINE FISHERIES

Fish bycatch research in these fisheries is 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 tows observed in 2009–10; Ramm 2012) and coverage has often been issue focused – e.g., monitoring of Hector's and Māui dolphin interactions and abundance for the Threat Management Plan – rather than representative. There are also practical and logistical problems with placing observers on smaller inshore vessels, and other options are being explored for 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 fisheries.

Electronic reporting and monitoring is being implemented in a phased manner across all New Zealand fisheries (including for the inshore). Some progress has been made with estimating the bycatch of undersized fish such as sub-legal-sized snapper (SNX), but there remain some issues to surmount before electronic monitoring can provide all the information required to estimate fish and invertebrate bycatch.

1.3.2 CHATHAM RISE (CHAT)

The Chatham Rise is an important region for all the major deepwater fisheries except for southern blue whiting. Total bycatch from deepwater fisheries has ranged from about 15 000 t to about 37 000 t, with generally decreasing amounts since 2000–01, but a small rise after 2011–12 (Figure 9.2). In each year since 1995–96 the hoki/hake/ling trawl fishery has been the main contributor to total bycatch in this area. Prior to that most of the bycatch from deepwater fisheries was attributed to the orange roughy fishery. The arrow squid and scampi fisheries also contributed substantially in some years.

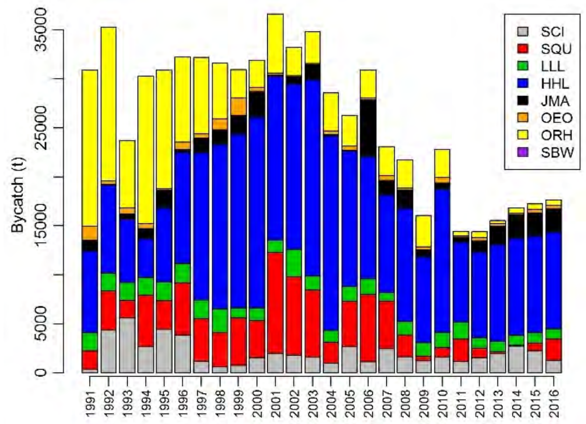

Figure 9.2: Estimated total annual bycatch (source fisheries shown by bar colouration) in Chatham Rise deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards on the Chatham Rise from deepwater fisheries has ranged from about 4000 t to about 18 000 t, with generally lower amounts since 2003–04 (Figure 9.3). In most years the largest contributor to discards by volume in this area was the hoki/hake/ling trawl fishery. Discards were relatively low in the orange roughy fishery compared to bycatch as a large part of the bycatch in that fishery, especially in earlier years, is likely to have been commercial species – oreos in particular.

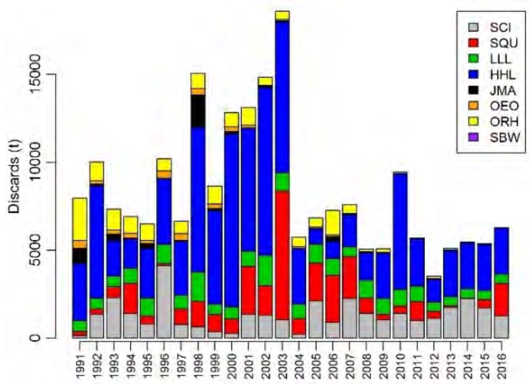

Figure 9.3: Estimated total annual discards (source fisheries shown by bar colouration) in Chatham Rise deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.3 SUBANTARCTIC (SUBA)

The subantarctic is an important region for all the major deepwater fisheries except for jack mackerel. Total bycatch from deepwater fisheries in the area has ranged from about 300 t to about 4100 t, with variable levels but generally lower since 2004–05 (Figure 9.4). In the past major contributors have been the ling longline, southern blue whiting, orange roughy, and hoki/hake/ling fisheries. Most recently the hoki/hake/ling (including longline) fisheries have been the greatest contributors.

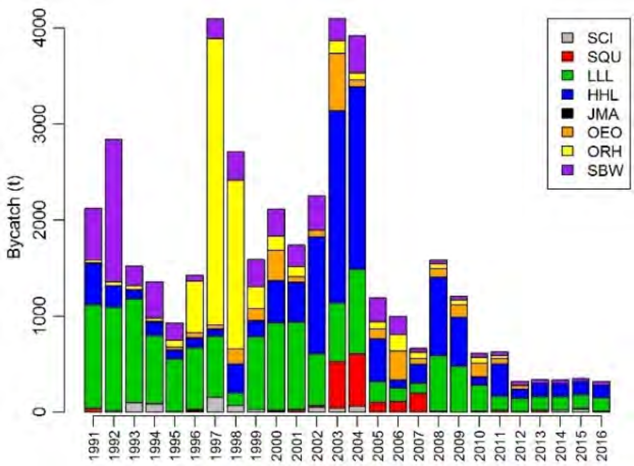

Figure 9.4: Estimated total annual bycatch (source fisheries shown by bar colouration) in subantarctic deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the subantarctic from deepwater fisheries has ranged from about 200 t to about 1600 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 target species. Currently total discards are mostly split between the southern blue whiting and ling longline fisheries.

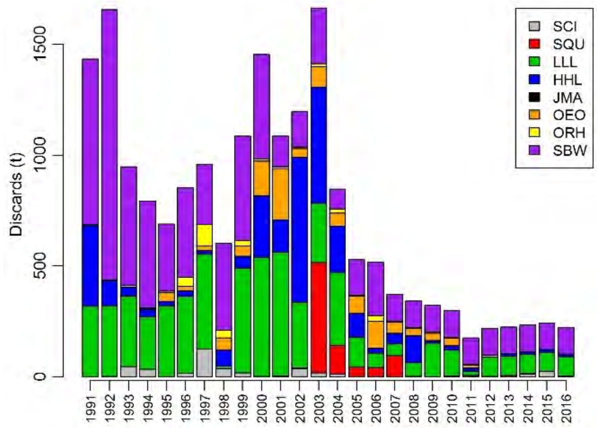

Figure 9.5: Estimated total annual discards (source fisheries shown by bar colouration) in subantarctic deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.4 STEWART-SNARES SHELF (STEW)

The Stewart-Snares Shelf is an important region for the jack mackerel, hoki/hake/ling and arrow squid trawl fisheries, with smaller fisheries also operating for oreo and orange roughy and ling (longline). Total bycatch in the Stewart-Snares Shelf area from deepwater fisheries has ranged from about 3000 t to about 32 000 t per year, with the lowest values in the mid-1990s, but falling levels since 2005–06 (Figure 9.6). The majority of this bycatch, in all years except for 1994–95, has been in the arrow squid fishery, with most of the remainder coming from the hoki/hake/ling fishery. In several years there has also been a notable contribution from the jack mackerel fishery.

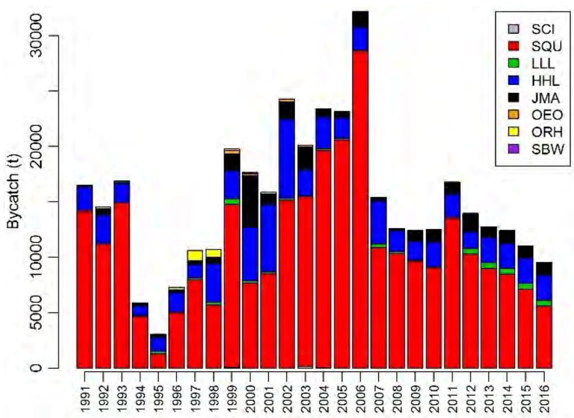

Figure 9.6: Estimated total annual bycatch (source fisheries shown by bar colouration) in Stewart-Snares Shelf deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Stewart-Snares Shelf area from deepwater fisheries has ranged from about 500 t to about 6000 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.

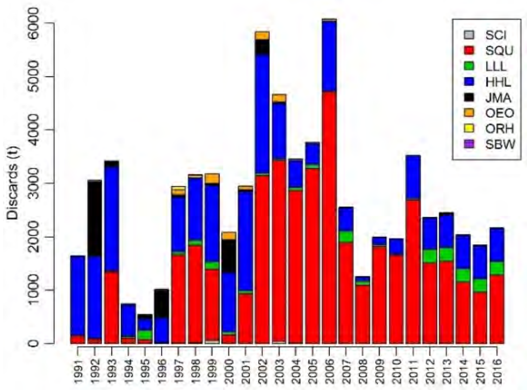

Figure 9.7: Estimated total annual discards (source fisheries shown by bar colouration) in Stewart-Snares Shelf deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.5 AUCKLAND ISLANDS (AK)

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 bycatch in the Auckland Islands area from deepwater fisheries has ranged from about 750 t to about 7500 t per year, but fluctuating between about 1200 t and 4000 t since 1997–98 (Figure 9.8). The main contributors to bycatch 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.

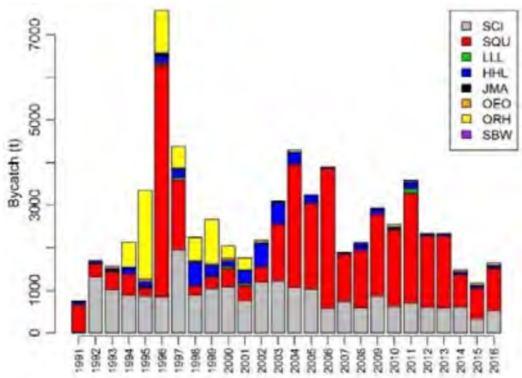

Figure 9.8: Estimated total annual bycatch (source fisheries shown by bar colouration) in Auckland Islands deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Auckland Islands area from deepwater 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). Although variable, current levels are typical of the last 15 years.

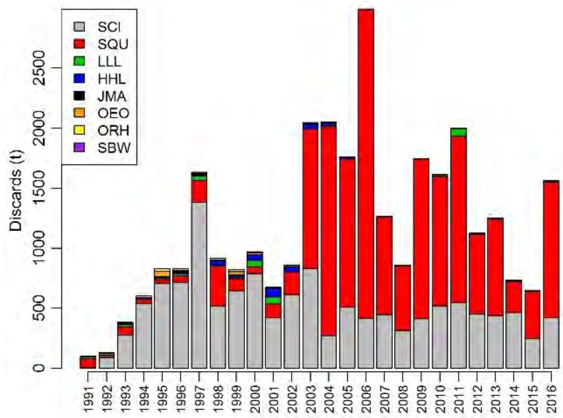

Figure 9.9: Estimated total annual discards (source fisheries shown by bar colouration) in Auckland Islands deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.6 PUYSEGUR (PUYS)

Most deepwater fisheries have operated at some time in the Puysegur area, with bycatch 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 bycatch in the area from deepwater fisheries has ranged from about 200 t to about 4500 t per year, with generally decreasing amounts since about 2000–01 (Figure 9.10). Annual bycatch is currently relatively low, and attributed mostly to the hoki/hake/ling fishery, with smaller contributions from the arrow squid and orange roughy fisheries.

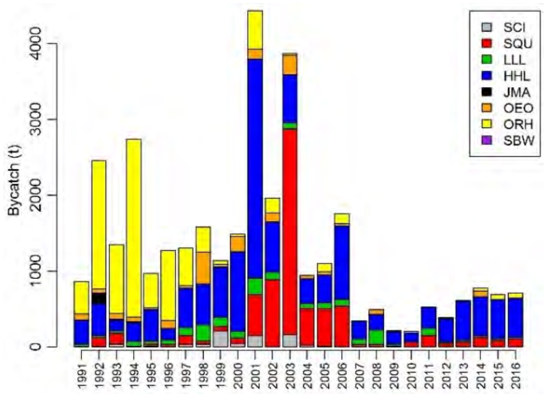

Figure 9.10: Estimated total annual bycatch (source fisheries shown by bar colouration) in Puysegur deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Puysegur area from deepwater fisheries has ranged from about 100 t to about 3500 t per year with 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 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.

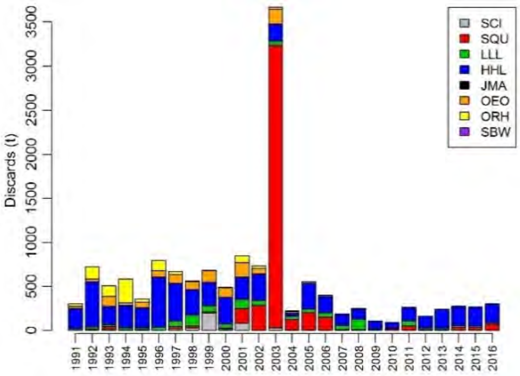

Figure 9.11: Estimated total annual discards (source fisheries shown by bar colouration) in Puysegur deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.7 WEST COAST SOUTH ISLAND (WCSI)

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, mainly before 2000, an orange roughy fishery. Currently most bycatch can be attributed to the hoki/hake/ling fishery. Total bycatch in the west coast South Island area from deepwater fisheries has ranged from about 2500 t to about 18 000 t per year, with generally decreasing amounts since about 1994–95 (Figure 9.12).

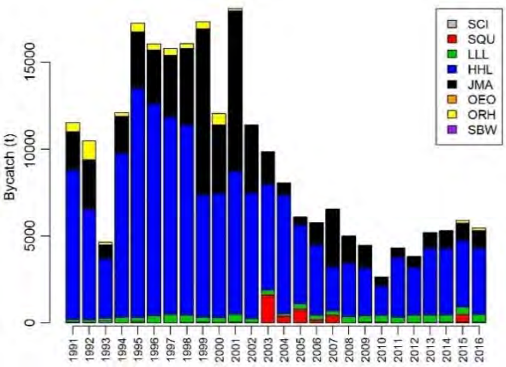

Figure 9.12: Estimated total annual bycatch (source fisheries shown by bar colouration) in west coast South Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the west coast South Island area from deepwater 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 to bycatch (Figure 9.13). Total discards have been below 2500 t per year since 2003–04, attributed mostly to the hoki/hake/ling fishery, with a small contribution from the ling longline fishery.

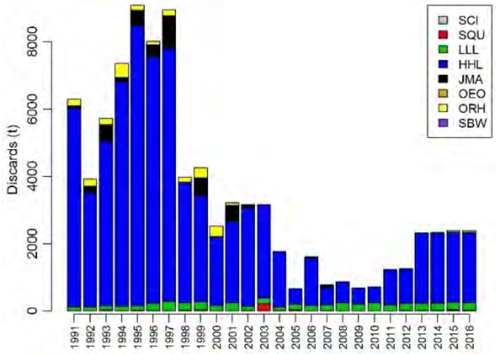

Figure 9.13: Estimated total annual discards (source fisheries shown by bar colouration) in west coast South Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.8 WEST COAST NORTH ISLAND (WCSI)

The dominant deepwater fishery on the west coast North Island region is currently the jack mackerel trawl fishery, with fisheries for orange roughy and arrow squid operating mainly before 2003–04. Total bycatch in the west coast North Island area from deepwater fisheries has ranged from about 1100 t to about 13 000 t per year, with generally decreasing amounts since 2003–04 (Figure 9.14). In most years almost all of the bycatch can be attributed to the jack mackerel fishery, but with substantial contributions from the orange roughy and arrow squid fisheries in one or two years.

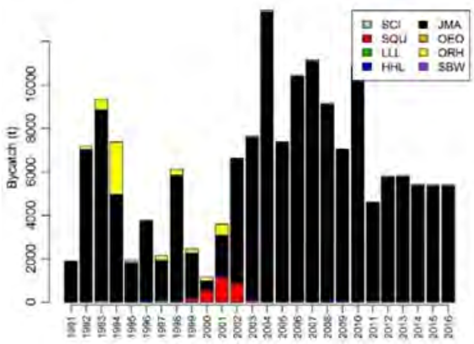

Figure 9.14: Estimated total annual bycatch (source fisheries shown by bar colouration) in west coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the west coast North Island area from deepwater 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 but still dominates in most years.

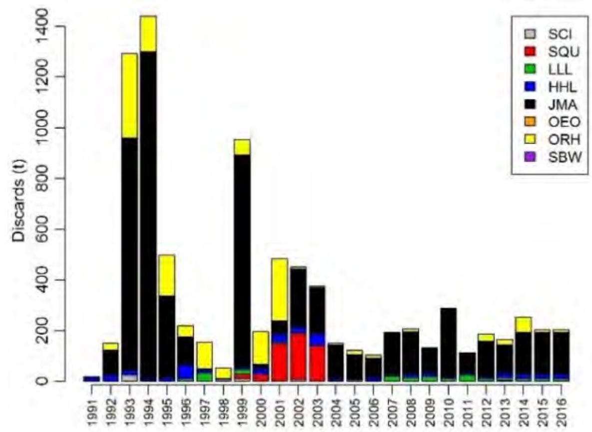

Figure 9.15: Estimated total annual discards (source fisheries shown by bar colouration) in west coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.9 NORTHLAND (NRTH)

Deepwater 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 bycatch in the area from deepwater fisheries has ranged from about 500 t to about 5000 t per year, but with generally stable levels of less than 1400 t per year since about 1998–99 (Figure 9.16). In most years bycatch 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 deepwater fisheries are minor in this area and currently total annual bycatch is less than 1000 t, split between the scampi and hoki/hake/ling fisheries.

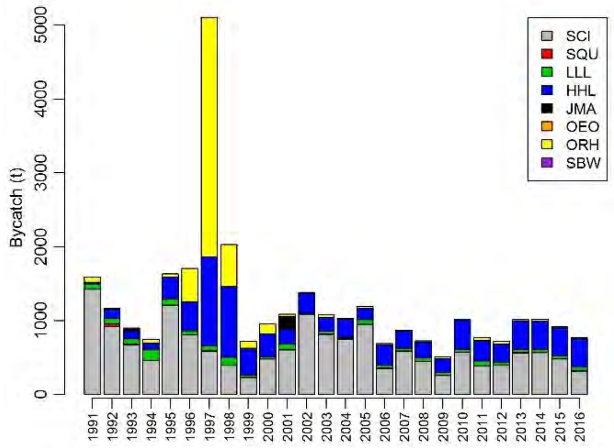

Figure 9.16: Estimated total annual bycatch (source fisheries shown by bar colouration) in Northland deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Northland area from deepwater fisheries have ranged from about 300 t to about 1000 t per year, with levels of about 300–500 t per year over the last several years (Figure 9.17). Discards in this area are dominated by the scampi fishery in all but two years in which the hoki/hake/ling and orange roughy fisheries contributed more.

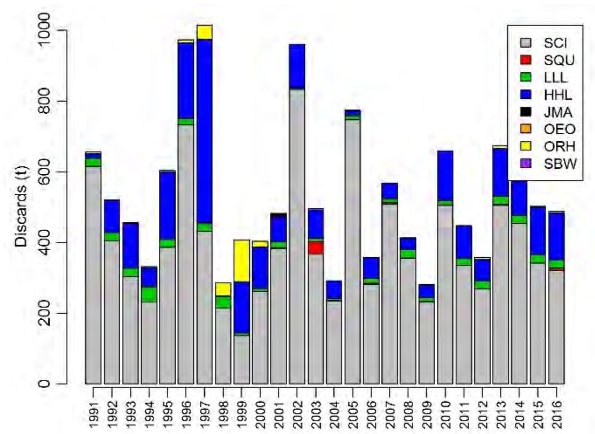

Figure 9.17: Estimated total annual discards (source fisheries shown by bar colouration) in Northland deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.10 EAST COAST NORTH ISLAND (EAST)

The main deepwater fisheries operating in the East Coast North Island have been the scampi, hoki/hake/ling, and orange roughy fisheries, and the ling longline fishery. Total bycatch in the area from deepwater fisheries has ranged from about 1000 t to about 7500 t per year, with generally decreasing levels since about 1997–98 (Figure 9.18). Most of the bycatch comes from the hoki/hake/ling and scampi fisheries, with larger contributions from the orange roughy fishery before 2004–05, and lower contributions from the ling longline fishery.

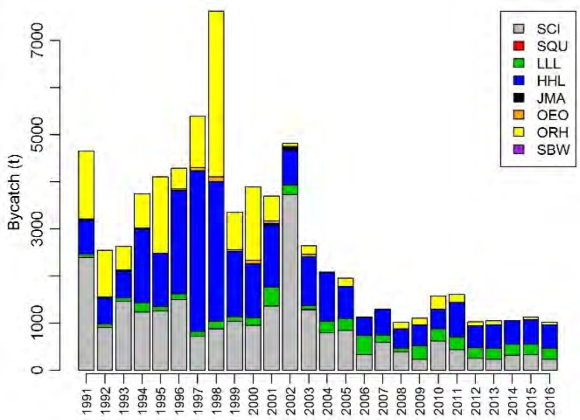

Figure 9.18: Estimated total annual bycatch (source fisheries shown by bar colouration) in East Coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the East Coast North Island area from deepwater fisheries has ranged from about 400 t to about 2800 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 bycatch 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 500 t, mostly associated with the scampi fishery.

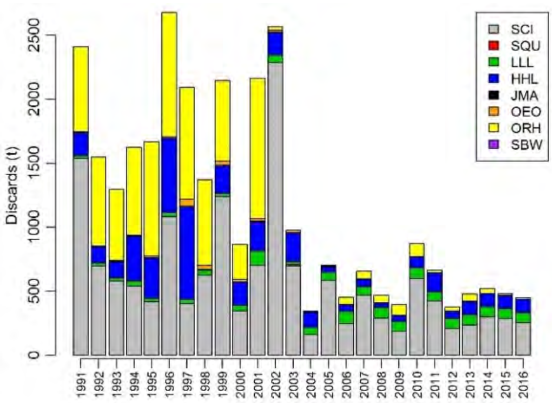

Figure 9.19: Estimated total annual discards (source fisheries shown by bar colouration) in East Coast North Island deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.11 COOK STRAIT (COOK)

The main fishery in the Cook Strait area has been the hoki/hake/ling fishery, with this fishery contributing the great majority of bycatch in most years. Total bycatch in the Cook Strait area from deepwater fisheries has ranged from about 400 t to about 5000 t per year, with generally decreasing levels since about 1996–97 (Figure 9.20). The orange roughy fishery operating on the fringes of this area also contributed substantially to total annual bycatch through the early 1990s. Currently total annual bycatch is less than 1000 t, almost all from the hoki/ hake/ling fishery.

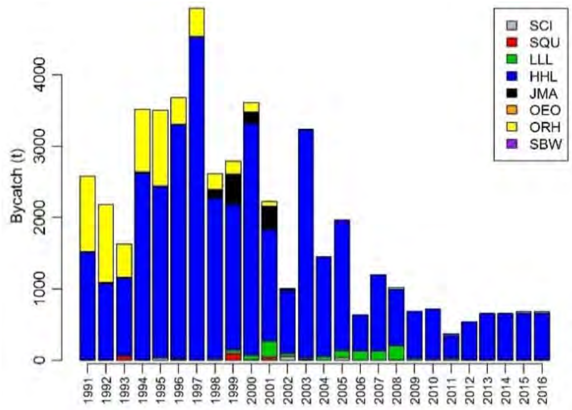

Figure 9.20: Estimated total annual bycatch (source fisheries shown by bar colouration) in Cook Strait deepwater fisheries. For fisheries abbreviations see Table 9.1.

Total discards in the Cook Strait area from deepwater fisheries has 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 t per year.

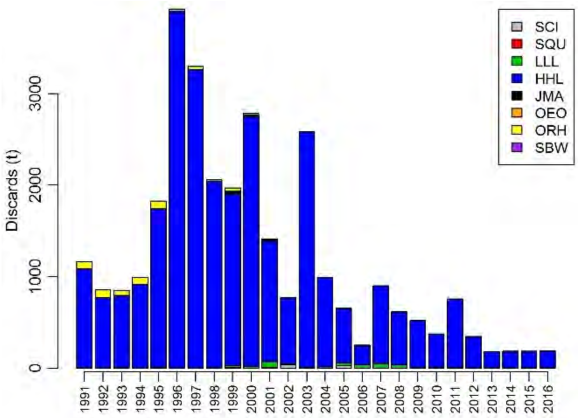

Figure 9.21: Estimated total annual discards (source fisheries shown by bar colouration) in Cook Strait deepwater fisheries. For fisheries abbreviations see Table 9.1.

1.3.12 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*)[1] and higher still after 2011–12, 90–97%, due to 100% coverage requirements for Foreign Charter Vessels (FCVs). 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. Observers covered the full size range of vessels operating in the fishery, although 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 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 bycatch and discards in this fishery (Anderson & Edwards, in prep) 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 bycatch 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.

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 bycatch (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.

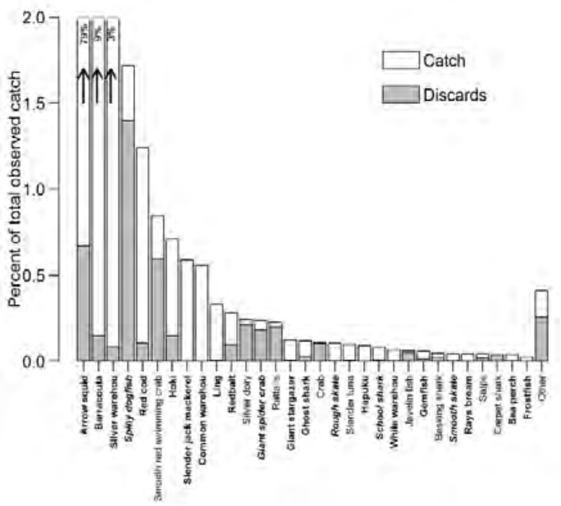

Figure 9.22: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch) in the observed portion of the arrow squid fishery, and the percentage discarded, 1 October 2001 to 30 September 2016 (Anderson & Edwards, in prep). The 'Other' category is the sum of all bycatch species representing less than 0.02% of the total catch. QMS species are shown in bold, Schedule 6 species are in italics.

[1] Ministry for Primary Industries (2012) Operational plan to manage the incidental capture of New Zealand sea lions in the Southern Squid Trawl Fishery (SQU6T). ISBN Online: 978-0-478-42003-6; ISSN Online: 2253-3923.

Total annual bycatch 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 large majority of the bycatch comprised QMS species, with less than 1000 t of non-QMS species and invertebrate species bycatch in most years.

TRENDS IN ESTIMATED BYCATCH BY SPECIES FROM THE ARROW SQUID TRAWL FISHERY

Anderson & Edwards (in prep) estimated the level of the main individual fish and invertebrate species bycatch in each fishing year from 2002–03 to 2015–16. The following conclusions were made:

- The most commonly caught bycatch species were barracouta (BAR), silver warehou (SWA), spiny dogfish (SPD), red cod (RCO), rattails (RAT), blue warehou (WAR), ling (LIN), and hoki (HOK) (Figure 9.24).
- Of the 30 bycatch species examined, 5 showed a significant decrease in catch over time and 2 an increase in catch.
- The species showing significant declines were barracouta, silver warehou, and spiny dogfish.
- The species showing significant increases were basking shark (BSK – although no catches have been recorded in the last three years) and hake (HAK).

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 (Figure 9.25). 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%).

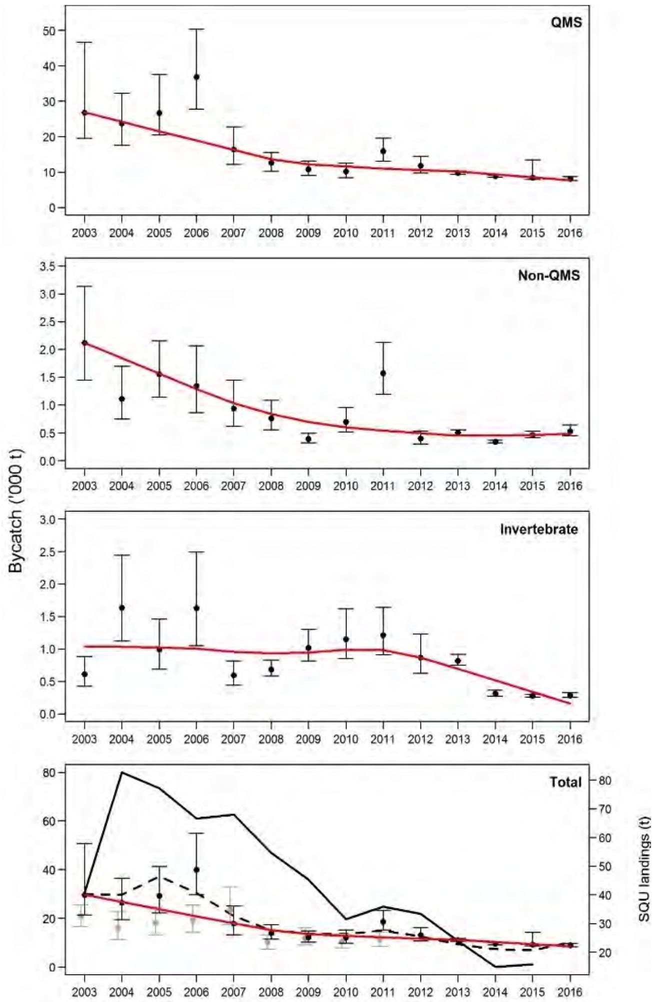

Figure 9.23: Annual estimates of bycatch in the arrow squid trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (from Anderson & Edwards, in prep). Also shown (in grey) are estimates of total bycatch calculated for 2002–03 to 2010–11 (Anderson 2013a). 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).

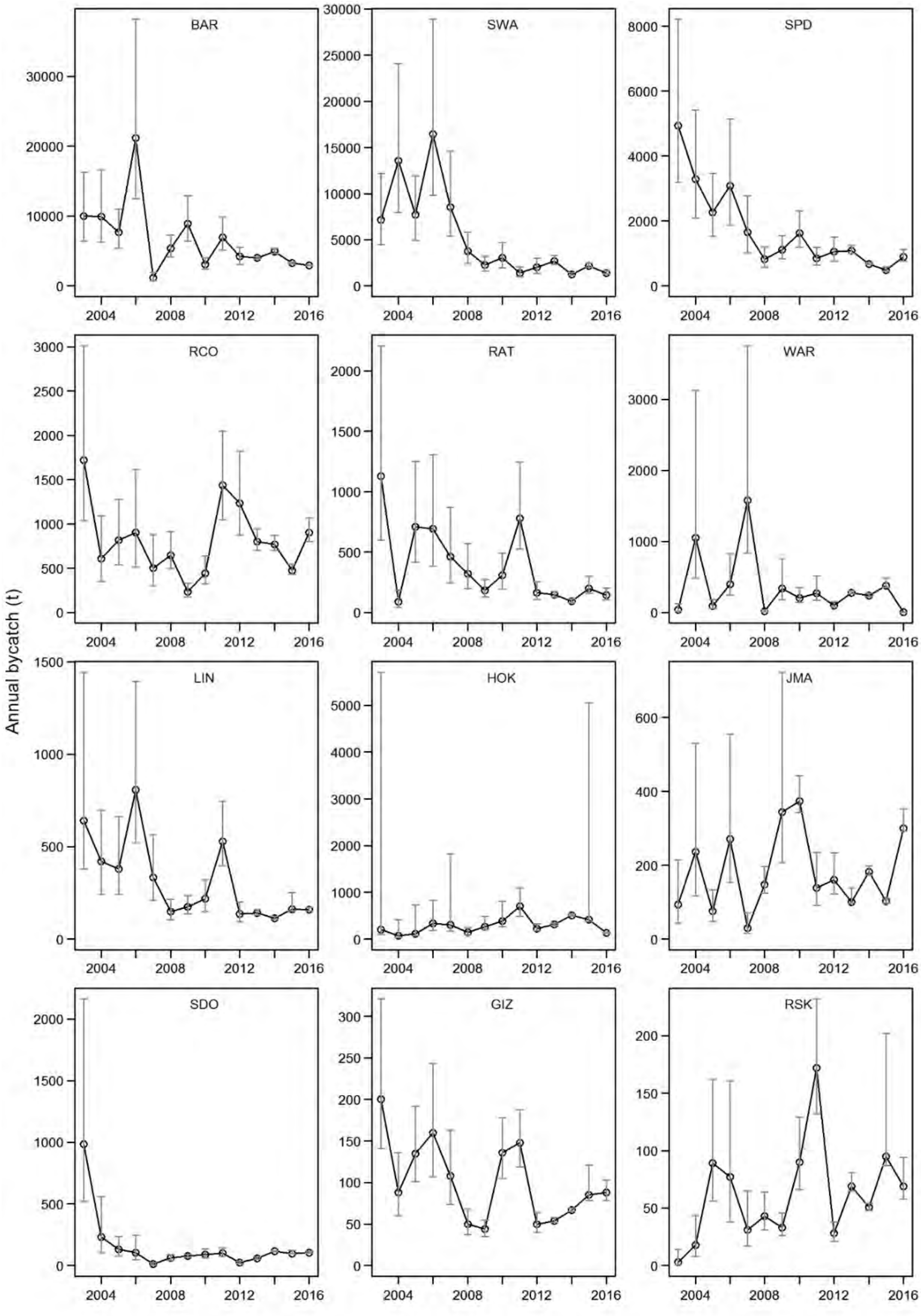

Figure 9.24: Annual bycatch estimates in the target arrow squid trawl fishery for the species that have the most bycatch between 2002–03 and 2015–16, with 95% c.i.s, in descending order of total catch. See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots (from Anderson & Edwards, in prep).

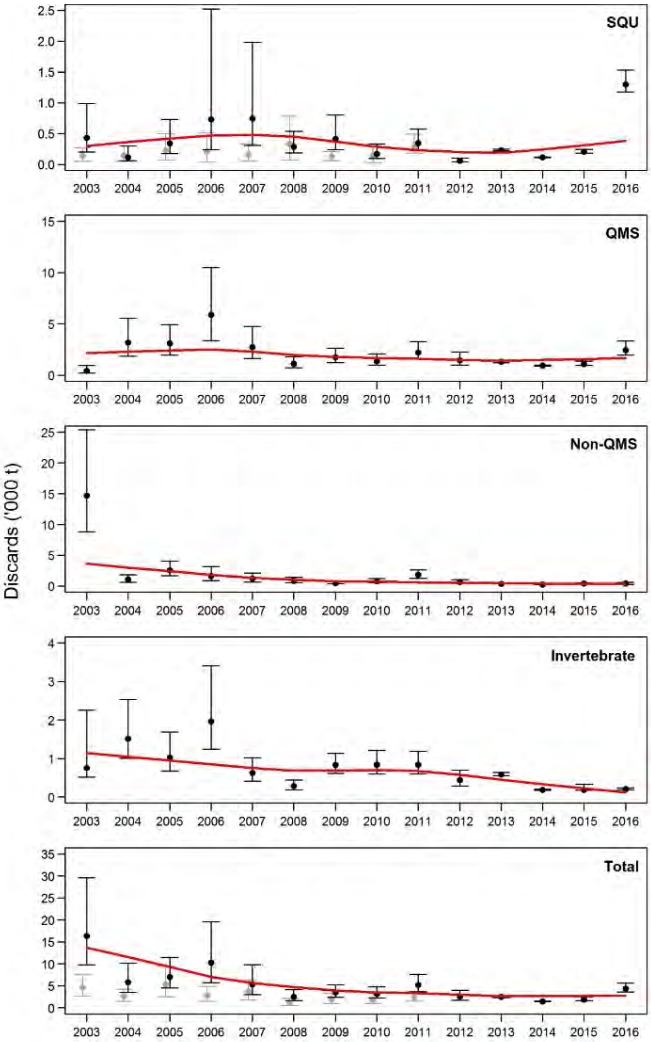

Figure 9.25: Annual estimates of discards in the arrow squid trawl fishery, for arrow squid (SQU), QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (from Anderson & Edwards, in prep). Also shown (in grey) are estimates of arrow squid and Total discards calculated 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.

1.3.13 HOKI/HAKE/LING TRAWL FISHERY

Earlier analyses were limited to the hoki target fishery but more recent work has expanded to cover bycatch and discards from hoki, hake and ling target fisheries combined; hoki nevertheless dominates this fishery, accounting for over 90% of the catch (Ballara & O'Driscoll 2015). Between 1990–91 and 2012–13, observer sampling levels have been highest in the west coast South Island, Chatham Rise, and Subantarctic areas, with lower levels in Cook Strait and Puysegur (these areas comprise the majority of the fishery in any year); little sampling has occurred outside of these main fishery areas. Higher levels of observer coverage were always achieved during the hoki spawning season (July to early September), but coverage outside of this period was variable and under-representative in some months in some years, especially in the subantarctic, Chatham Rise and Puysegur fisheries, and inside the 25 n.mile line on the WCSI and in Cook Strait.

Hoki, hake and ling accounted for 91% (82%, 5.5% and 3.4%, respectively) of the total observed catch from trawls targeting hoki, hake, and ling between 1990–91 and 2012–13. The remaining 9% comprised a large range of species, in particular silver warehou (1.4%), javelinfish (1.4%), rattails (1.1%), and spiny dogfish (0.9%) (Figure 9.26). In total, over 470 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 spiny dogfish and basking shark, accounted for much of the non-commercial catch. Echinoderms, squids, crustaceans, and other unidentified invertebrates were also well represented in the bycatch of this fishery.

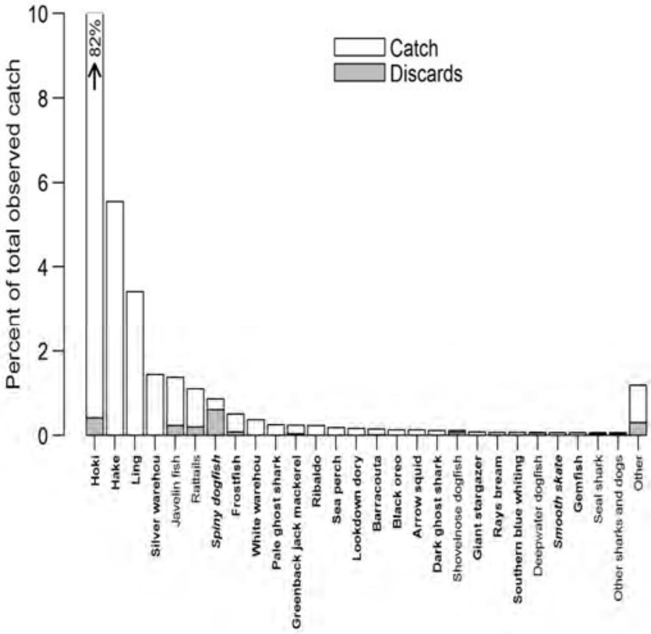

Figure 9.26: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch) in the observed portion of the hoki/hake/ling fishery (1990–91 to 2012–13), and the percentage discarded. QMS species are shown in bold (from Ballara & O'Driscoll 2015).

Total bycatch in the hoki, hake, and ling fishery between 1990–91 and 2012–13 was 12 000–38 000 t per year (compared to the combined total landed catch of hoki, hake, and ling of about 100 000–300 000 t). Overall, total bycatch increased during the 1990s to a peak in the early 2000s, then declined slowly. Annual bycatch for the 1990–01 to 2012–13 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 bycatch; invertebrate catch was less than 1000 t in most years, but peaked at about 1500 t in 2001–02 (Figure 9.27).

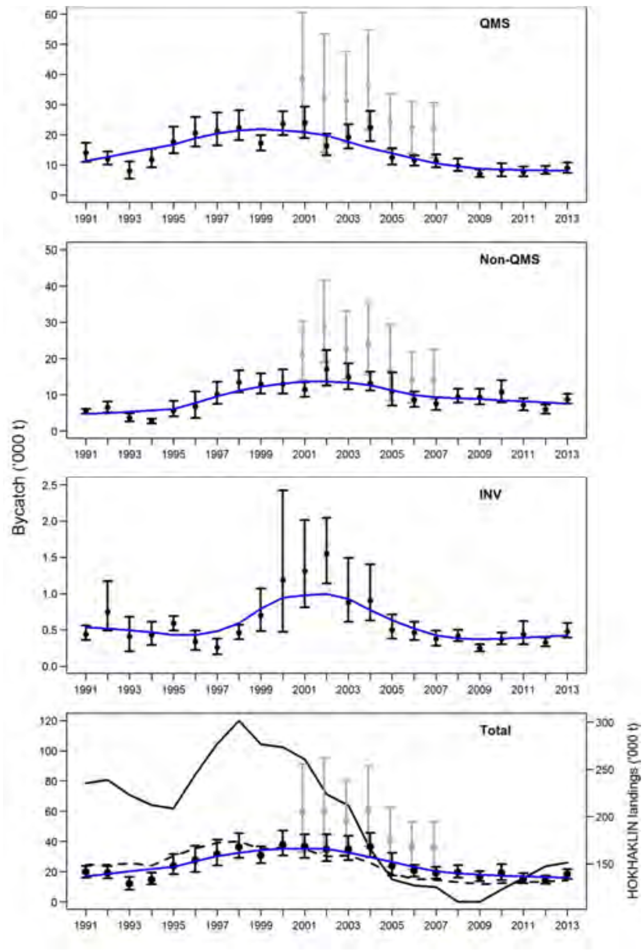

Figure 9.27: Annual estimates of fish bycatch in the target hoki, hake and ling trawl fishery, calculated for QMS species, non-QMS species, invertebrates, and overall for 1990–91 to 2012–13. Also shown (in grey) are earlier estimates of bycatch (Ballara et al. 2010). Error bars indicate 95% confidence intervals. The blue 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 hoki, hake, or ling (Ministry for Primary Industries 2014) and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

TRENDS IN BYCATCH BY SPECIES FROM THE HOKI, HAKE, AND LING TRAWL FISHERY

Ballara (2015) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2012–13. The following conclusions were made:

- The most commonly caught bycatch species were silver warehou (SWA), javelinfish (JAV), and unspecified rattails (Macrouridae, RAT).
- Of the 340 bycatch species examined, 40 had a decrease in catch over time and 19 an increase in catch.

- The species showing the greatest decline were skates (SKA), combined jack mackerel species (JMA, JMM, and JMN), and dogfishes (*Etmopterus* spp., ETM) (Figure 9.28). 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 Tam 'o Shanter urchins (Echinothurioida, TAM), umbrella octopus (*Opisthoteuthis* spp., OPI), and floppy tubular sponge (*Hyalascus* sp., HYA) (Figure 9.28).

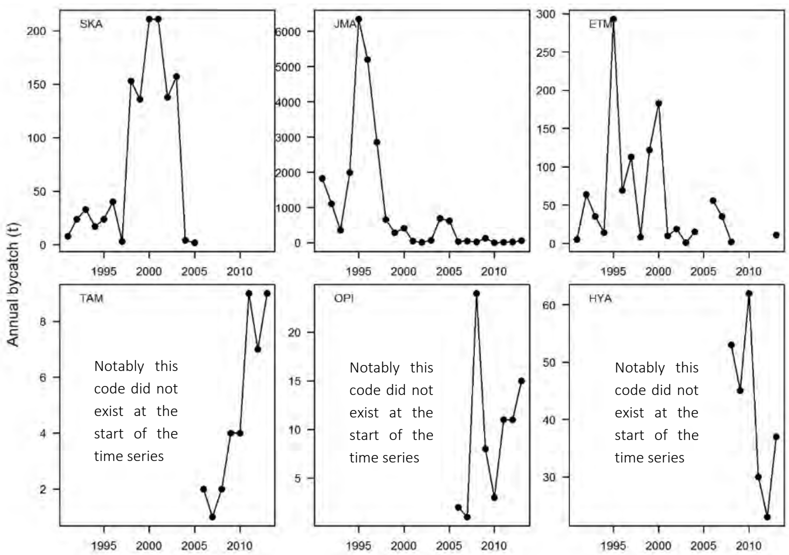

Figure 9.28: Annual bycatch estimates in the hoki, hake, and ling trawl fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2012–13. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1), and may be area-specific (see text above). See text above for species codes.

Total annual discard estimates for 1990–91 to 2012–13 were 3500–17 000 t per year with the main species observed discarded including spiny dogfish, rattails, javelinfish, and hoki. Estimated annual discards of the target species combined decreased substantially from about 5000–10 000 t before 1997–98 to less than 2000 t subsequently. Up until 2002–03 there was no obvious trend in total discards, but discards decreased markedly after this date (Figure 9.29). 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).

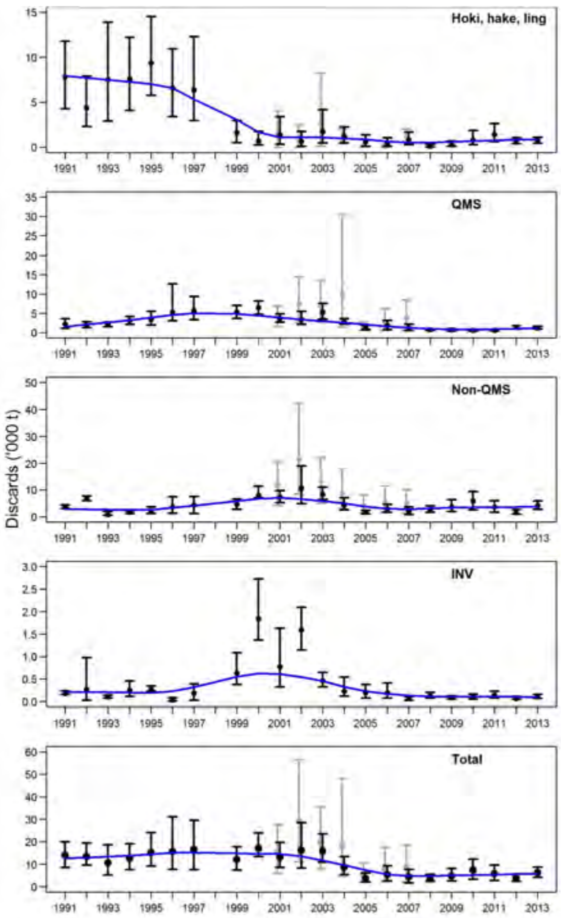

Figure 9.29: Annual estimates of discards in the hoki, hake or ling target trawl fishery, for hoki, hake, or ling target (HOKHAKLIN), QMS species, non-QMS species, invertebrates (INV), and overall for 1990–91 to 2012–13. Also shown (in grey) are earlier estimates of discards (Ballara et al. 2010). Error bars indicate 95% confidence intervals. The blue lines show the fit of a locally weighted polynomial regression to annual discards.

1.3.14 JACK MACKEREL TRAWL FISHERY

Estimates of annual bycatch in this fishery are available for fishing years up to 2013–14, with the most recent analysis focusing on the 2002–03 to 2013–14 period (Anderson et al. 2017b). Both the ratio and the statistical model method of estimation were used in this analysis, with comparable results overall although the statistical model provided tighter confidence intervals. The annual level of observer coverage in this fishery was 8–30% of the target fishery catch before 2007–08 but rapidly increased to be 80–95% after 2010–11. This elevated level of coverage was due to a commitment by MPI to full observer coverage on foreign charter vessels, which have historically taken a large part of the catch in this fishery (Ministry for Primary Industries 2013b). Observer effort in each year has generally been focused on the main fishery, off the west coasts of the North and South Islands, with some additional coverage on the Stewart/Snares Shelf and Chatham Rise fisheries. This was variable, however, and in 2003–04 and 2004–05 there were only 12 trawls observed outside of 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 although there were lengthy periods in some years when commercial fishing effort was not observed, coverage has been well matched to the main fishing periods for all years combined.

Jack mackerel species comprised 75% of the total observed catch from all trawls targeting jack mackerel from 2002–03 to 2012–13. The remaining 30% mostly comprised other QMS species; especially barracouta (13%), blue mackerel (3.4%), frostfish (3.4%), and redbait (2.5%) (Figure 9.30). Overall about 320 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[2] (which entered the QMS in October 2004), comprising about 0.3% of the total catch. Of the invertebrates, only molluscs (mostly arrow squid) were observed caught in substantial amounts (about 500 t) and these were mostly retained. Lesser amounts of cnidarians, sponges, and echinoderms were observed caught (about 13 t in total), and almost all were discarded.

Total bycatch in the jack mackerel trawl fishery from 2002–03 to 2013–14 was 6600–18 800 t. Estimates of total bycatch for 1990–91 to 2003–04 from earlier projects were 5400–15 500 t. Bycatch has mainly comprised QMS species, especially in more recent years (Figure 9.31).

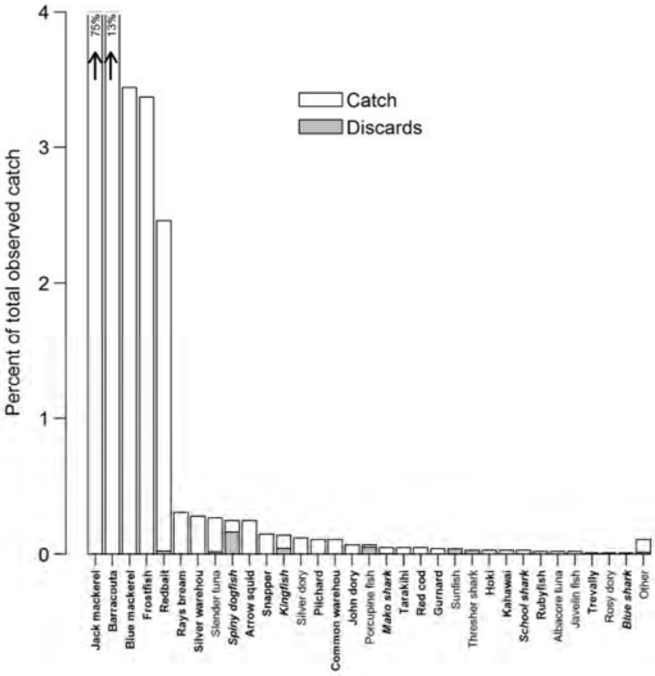

Figure 9.30: Percentage of the total catch contributed by the main bycatch species (those representing 0.01% or more of the total catch) in the observed portion of the jack mackerel trawl fishery between 2002–03 and 2013–14, and the percentage discarded. The 'Other' category is the sum of all bycatch 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).

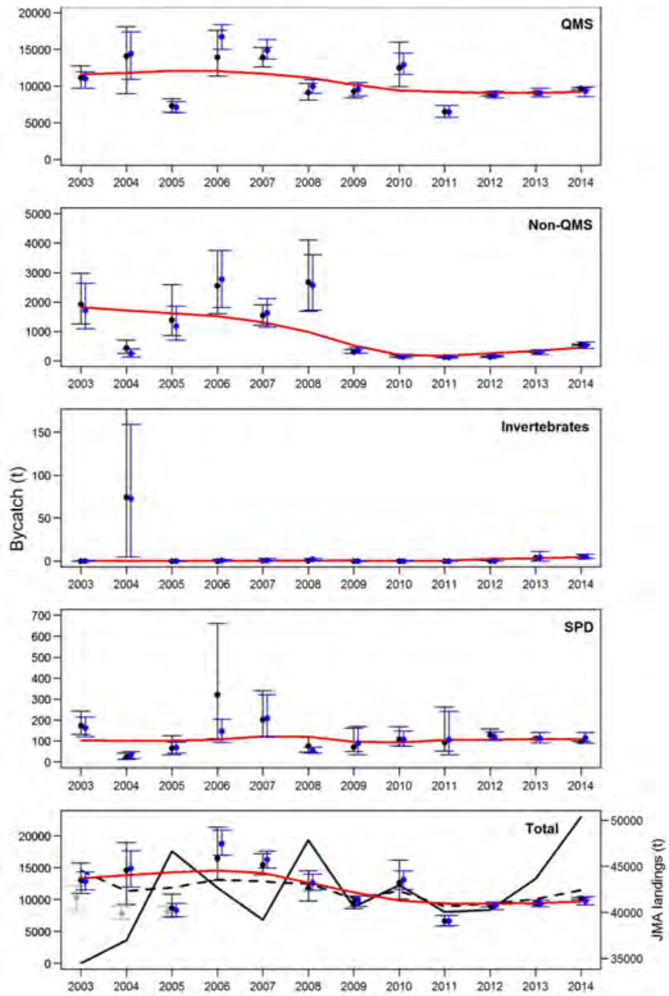

Figure 9.31: Annual estimates of bycatch in the jack mackerel trawl fishery, for QMS species, non-QMS species, invertebrates, and overall for 2002–03 to 2013–14: black dots, ratio method; blue dots, statistical model method. Also shown (in grey) are earlier estimates of total bycatch calculated for 2002–03 to 2004–05 (from Anderson 2007). 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 jack mackerels, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

TRENDS IN BYCATCH BY SPECIES FROM THE JACK MACKEREL TRAWL FISHERY

The report of Anderson et al. 2017b estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2013–14. The following conclusions were made:

[2] Notably it is legal to discard spiny dogfish under Schedule 6 of the Fisheries Act.

- The most commonly caught bycatch species were barracouta (BAR), blue mackerel (*Scomber australasicus*, EMA), and frostfish (*Lepidopus caudatus*, FRO).
- Of the 45 bycatch species examined, 13 showed a significant decrease in catch over time and 8 showed an increase.
- Species with significant declines included dark ghost shark (GSH), capro dory (*Capromimus abbreviatus*, CDO), hoki (HOK), and leatherjacket (*Meuschenia scaber*, LEA) (Figure 9.32).
- Species showing significant increases included albacore tuna (*Thunnus alalunga*, ALB), kahawai (*Arripis trutta*, ATT), and slender tuna (*Allothunnus fallai*, STU) (Figure 9.32).

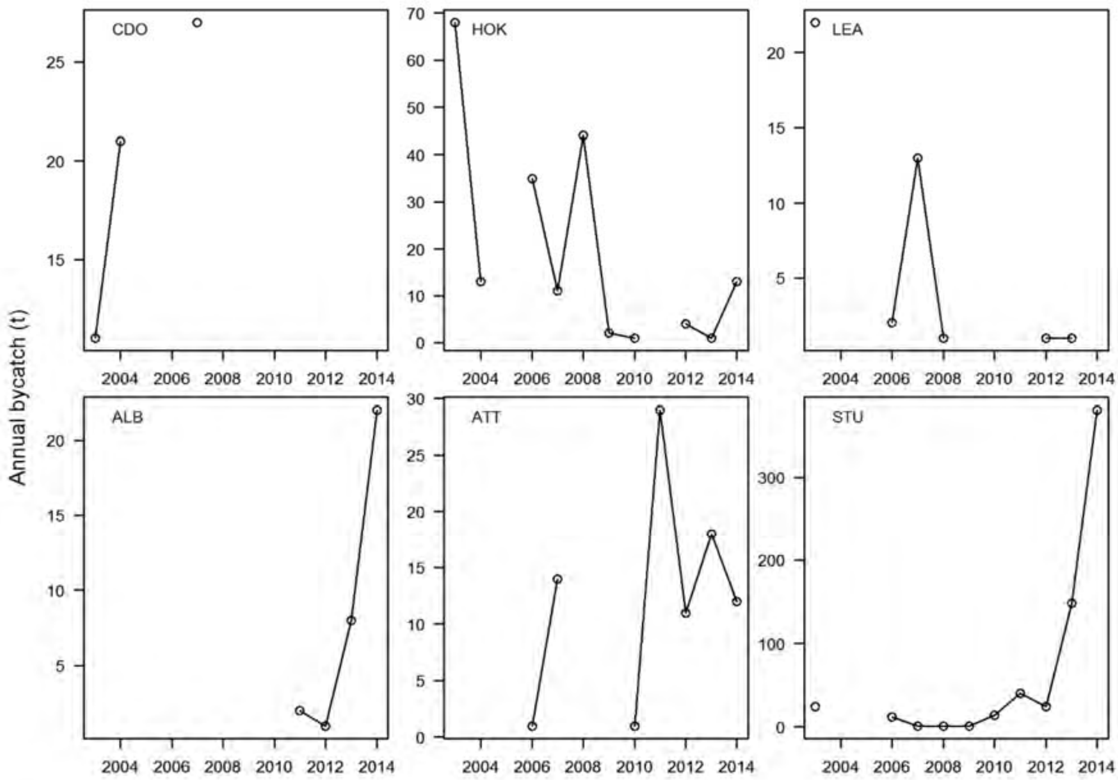

Figure 9.32: Annual bycatch estimates in the jack mackerel trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 2002–03 and 2013–14. See text above for explanation of the species codes. Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years (no data point = no observed catch).

Total annual discards varied between about 80 t and 400 t between 2002–03 and 2013–14, with no trend during this time. This compares with a much higher level in 1997–98 (1850 t), when annual discards were at their greatest. Annual discards of the target species were about 200–400 t prior to 1998–99 but thereafter decreased to about 10 t, mainly due to the absence of recorded losses of large quantities of fish through rips in the net or intentional releases of fish during landing. Discards comprised roughly equal amounts of QMS and non-QMS, although the ratio varied considerably from year to year (Figure 9.33).

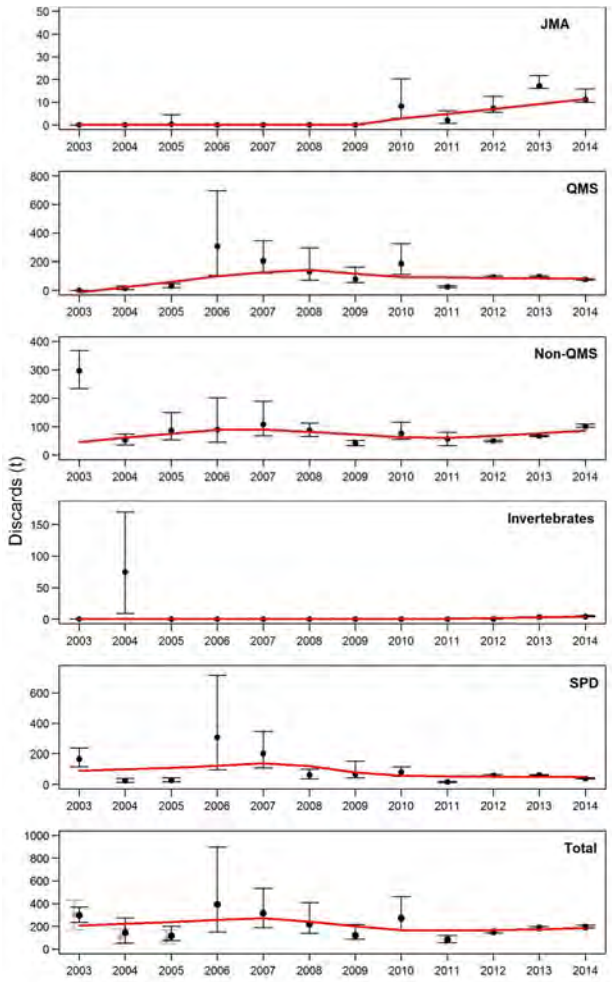

Figure 9.33: Annual estimates of discards in the jack mackerel trawl fishery, for jack mackerels (JMA), QMS species, non-QMS species, invertebrates, spiny dogfish (SPD), and overall (Total) for 2002–03 to 2013–14. Also shown (in grey) are earlier estimates of total discards calculated for 2002–03 to 2004–05 (from Anderson 2007). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

1.3.15 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.

The level of observer coverage represented was 22–53% of the target fishery catch from 2002–03 to 2006–07 and similar 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 were 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.

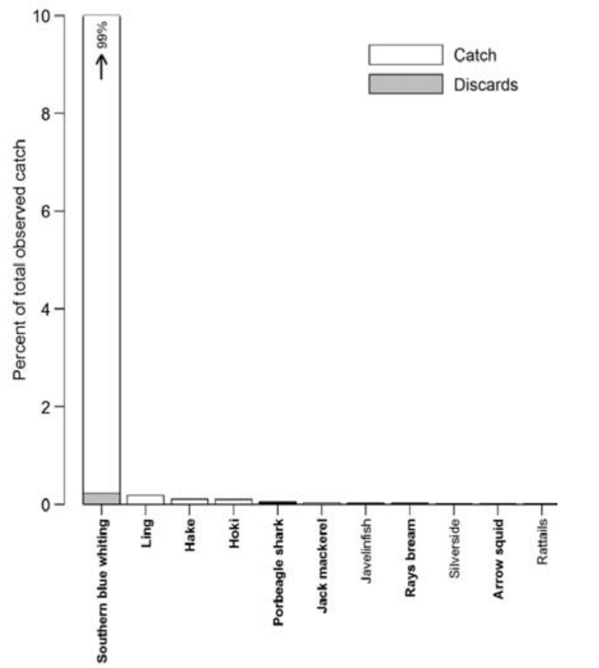

Figure 9.34: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch) in the observed portion of the southern blue whiting fishery, 2002–03 to 2006–07, and the percentage discarded. QMS species are shown in bold.

Total annual bycatch estimates 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 effort and the catch was exceptionally high.

TRENDS IN BYCATCH BY SPECIES FROM THE SOUTHERN BLUE WHITING TRAWL FISHERY

Ballara (2015) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2012–13. The following conclusions were made:

- The most commonly caught bycatch species were ling (*Genypterus blacodes*, LIN), hoki (*Macruronus novaezelandiae*, HOK), and hake (*Merluccius australis*, HAK).
- Of the 65 bycatch species examined, 4 had a decrease in catch over time and 11 had an increase in catch.
- The species showing the greatest decline were warty squid (*Onykia* spp., WSQ), lookdown dory (*Cyttus traversi*, LDO) and arrow squid (*Nototodarus sloanii* and *N. gouldi*, SQU) (Figure 9.36).
- The species showing the greatest increase were ray's bream (*Brama brama*, RBM), pale ghost shark (*Hydrolagus bemisi*, GSP), and opah (*Lampris immaculatus,* PAH) (Figure 9.36).

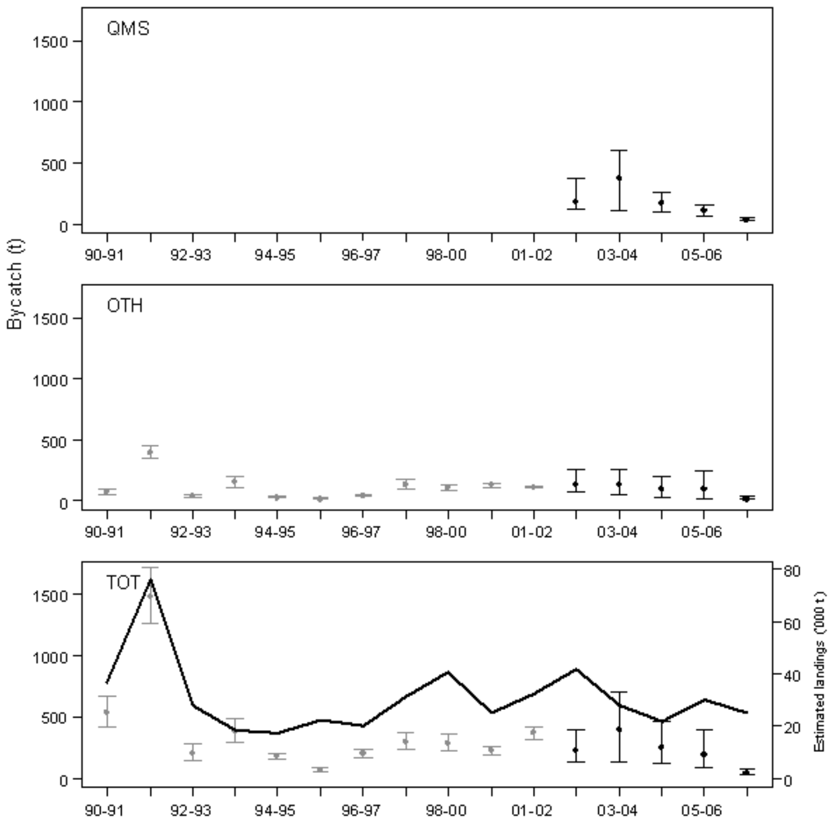

Figure 9.35: Annual estimates of fish bycatch in the southern blue whiting trawl fishery, calculated for QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black). Also shown (in grey) are estimates of bycatch in each category (excluding QMS) for 1990–91 to 2001–02 (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 between 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).

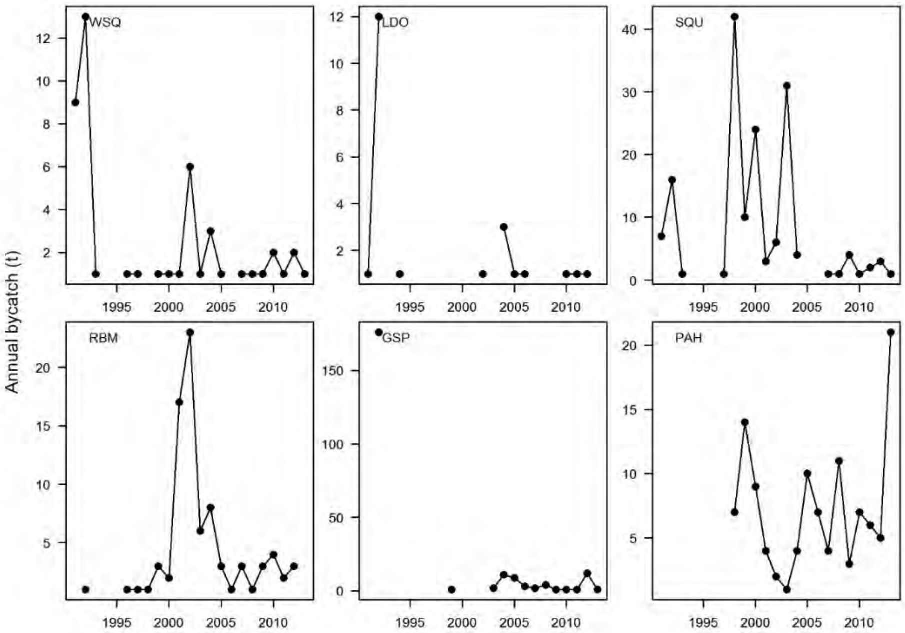

Figure 9.36: Annual bycatch estimates in the southern blue whiting trawl fishery for the species that had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2012–13. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1), and may be area-specific (see text above). See text above for species codes.

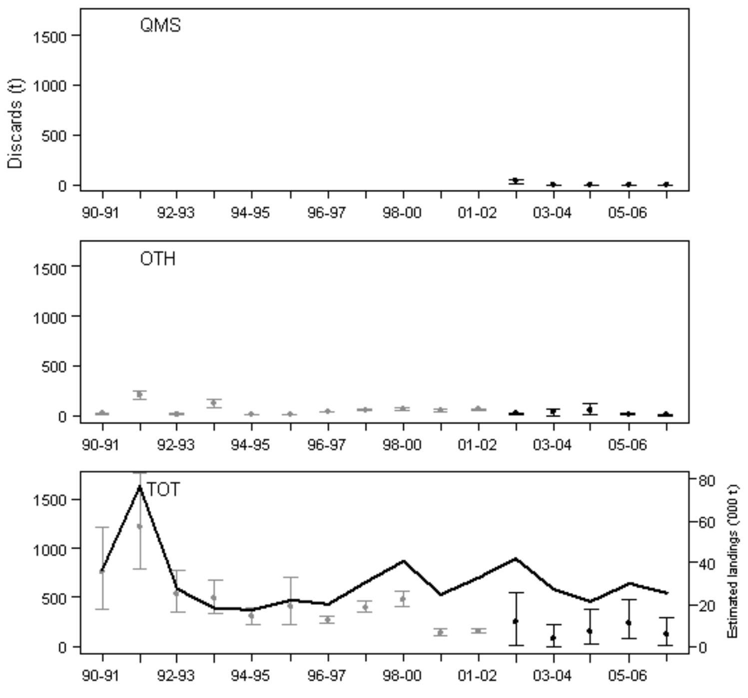

Figure 9.37: Annual estimates of fish discards in the southern blue whiting trawl fishery, calculated for the target species (SBW), QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black). Also shown (in grey) are estimates of discards in each category (excluding 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).

Total annual discard estimates from 2002–03 to 2006–07 were 90–250 t per year. 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 9.37). 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.

1.3.16 ORANGE ROUGHY TRAWL FISHERY

The most recent 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).

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 (which is the most widespread New Zealand fishery). Some undersampling occurred of smaller vessels, on the east coast fisheries in QMAs ORH 2A, ORH 2B and ORH 3A (where only small vessels are allowed to operate), and oversampling occurred of fisheries outside of 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 deepwater 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. 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).

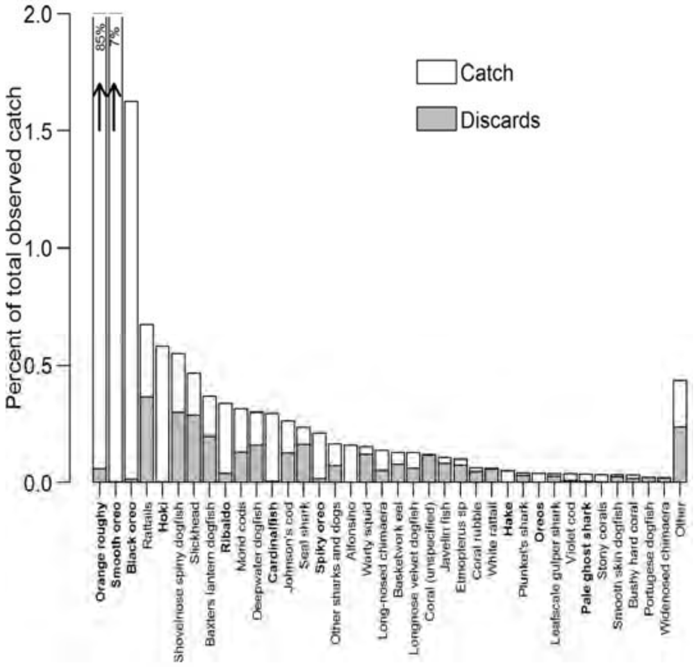

Figure 9.38: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch) in the observed portion of the target orange roughy trawl fishery for fishing years from 2001–02 to 2014–15, and the percentage discarded. The 'Other' category is the 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.

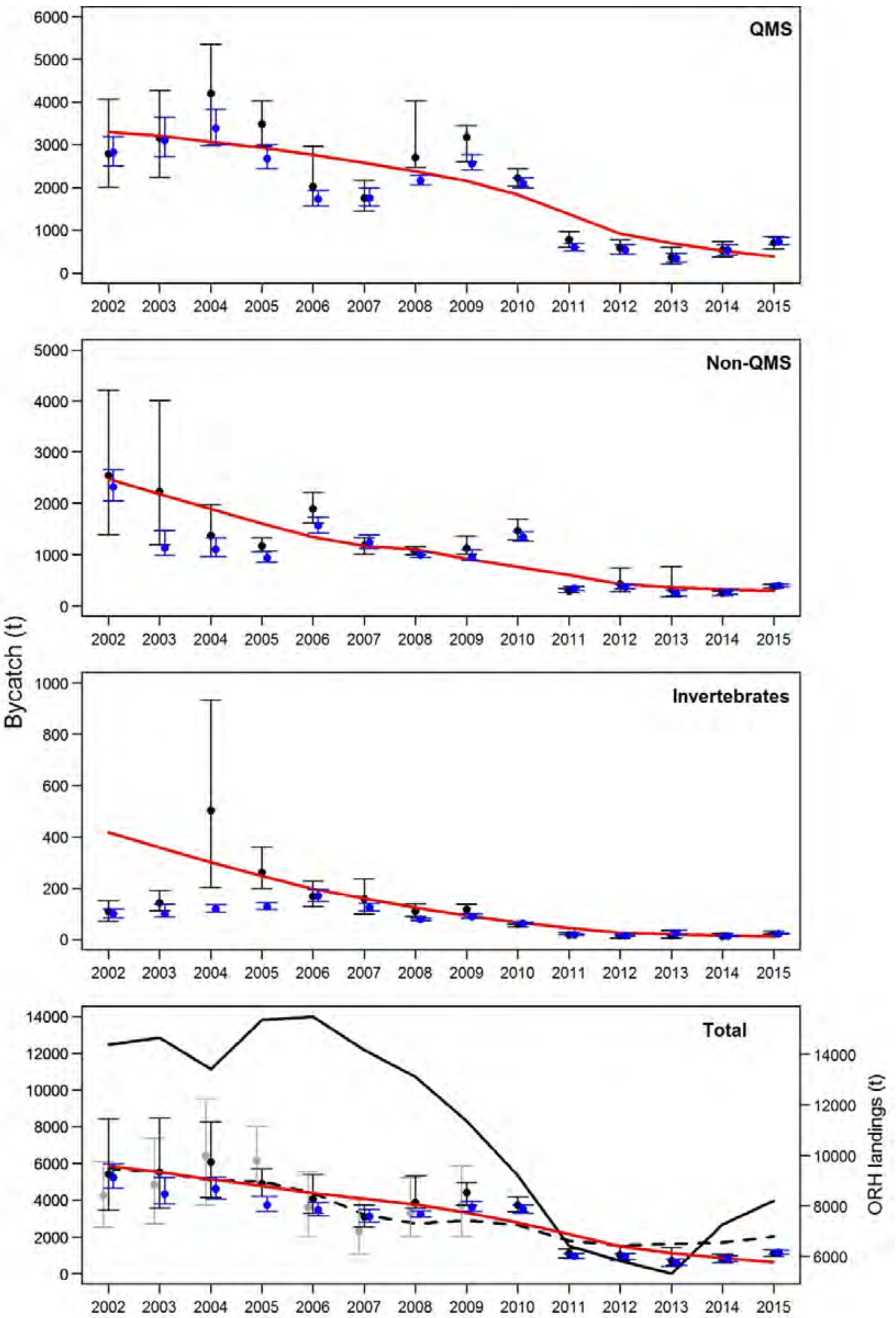

Figure 9.39: Annual estimates of bycatch (t) in the target orange roughy trawl fishery, species categories for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method. Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). 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.

TRENDS IN BYCATCH BY SPECIES FROM THE ORANGE ROUGHY TRAWL FISHERY

Anderson (2017) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2013–14. The following conclusions were made:

- The most commonly caught bycatch species were smooth oreo (*Pseudocyttus maculatus*, SSO), black oreo (*Allocyttus niger*, BOE), and black cardinalfish (*Epigonus telescopus*, CDL).
- Of the 211 bycatch species examined, 16 showed a significant decrease in catch over time and 3 showed a significant increase in catch.
- The species showing the greatest decline were black oreo, black cardinalfish, and *Beryx* spp., BYX (Figure 9.40).
- The species showing the greatest increase were morid cods (Moridae, MOD), longnose velvet dogfish (*Centroscymnus crepidater*, CYP), and bushy hard coral (*Goniocorella dumosa*, GDU; a species whose code did not exist before 2005–06) (Figure 9.40).

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 9.41). 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). Improved fishing techniques and gear have substantially lowered the level of discards in these categories in more recent times.

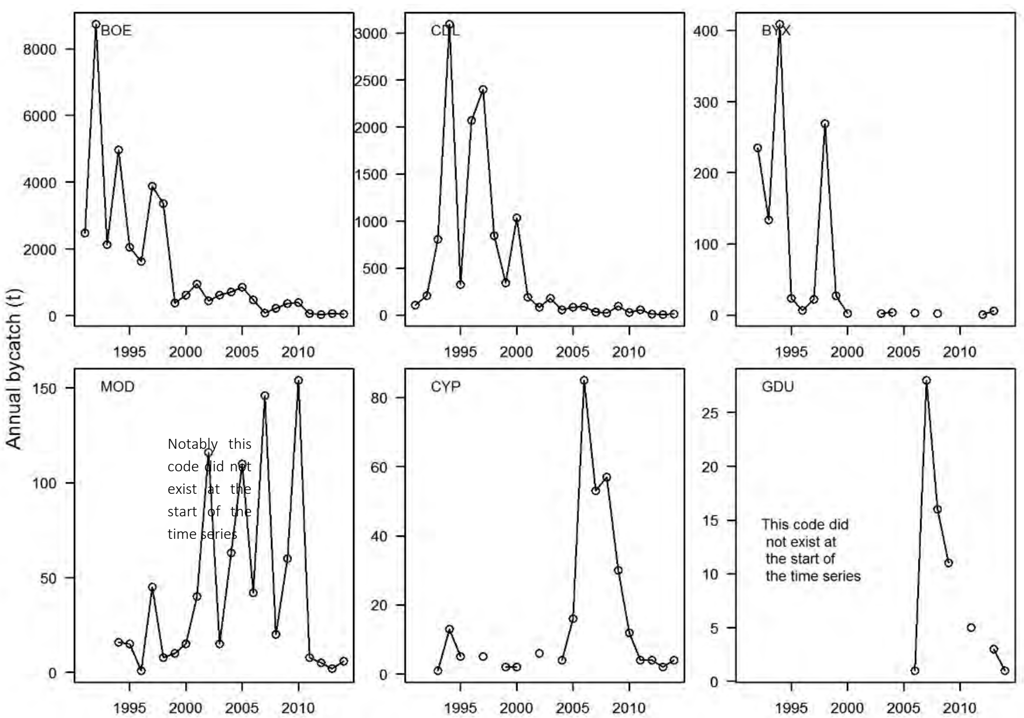

Figure 9.40: Annual bycatch estimates in the orange roughy trawl fishery for the species that have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2013–14. See text above for explanation of the species codes. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1). Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

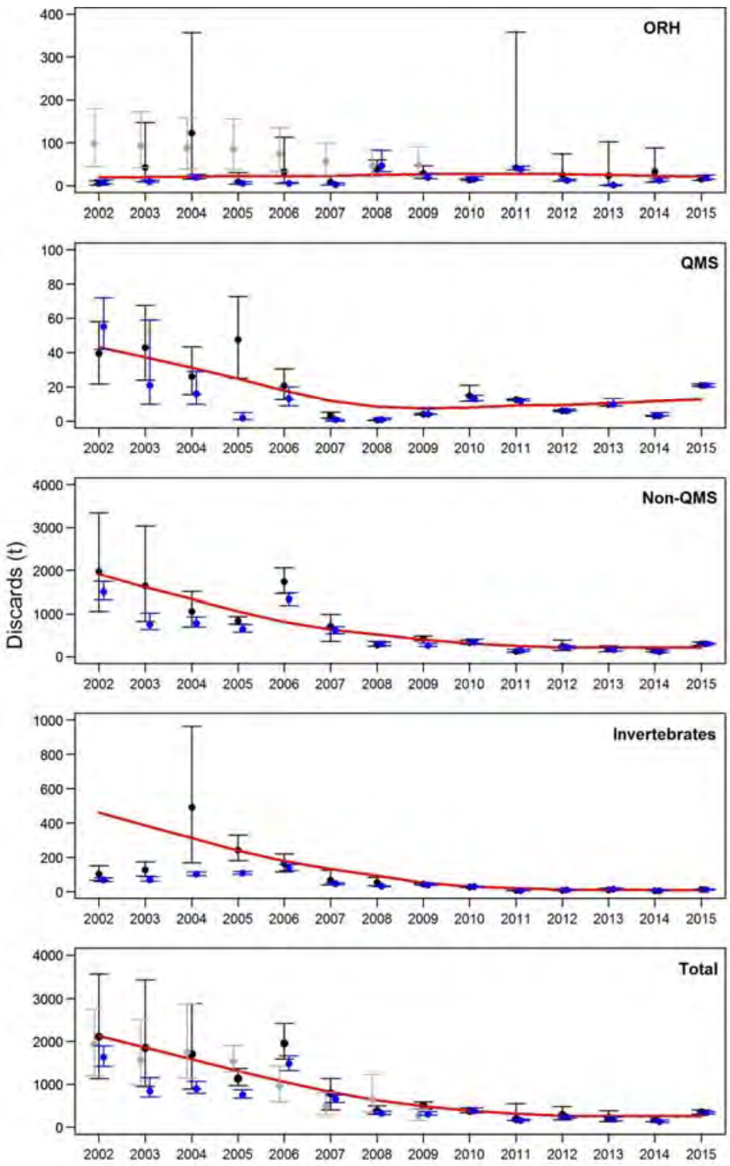

Figure 9.41: Annual estimates of discards (t) in the target orange roughy trawl fishery, for species categories for 2001–02 to 2014–15. Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

1.3.17 OREO TRAWL FISHERY

The most recent 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).

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 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.

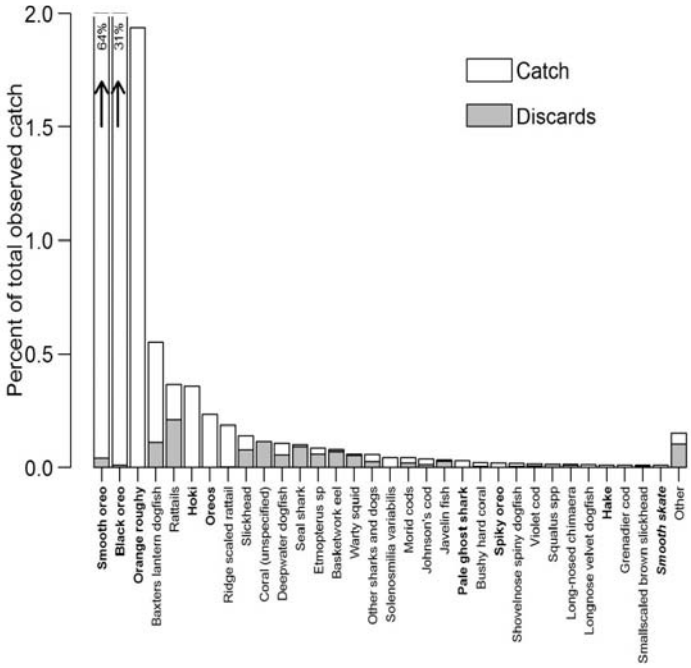

Figure 9.42: Percentage of the total catch contributed by the main bycatch species (those representing 0.01% or more of the total catch) in the observed portion of the oreo trawl fishery between 2001–02 and 2014–15, and the percentage discarded. The 'Other' category is the sum of all bycatch species 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)

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.

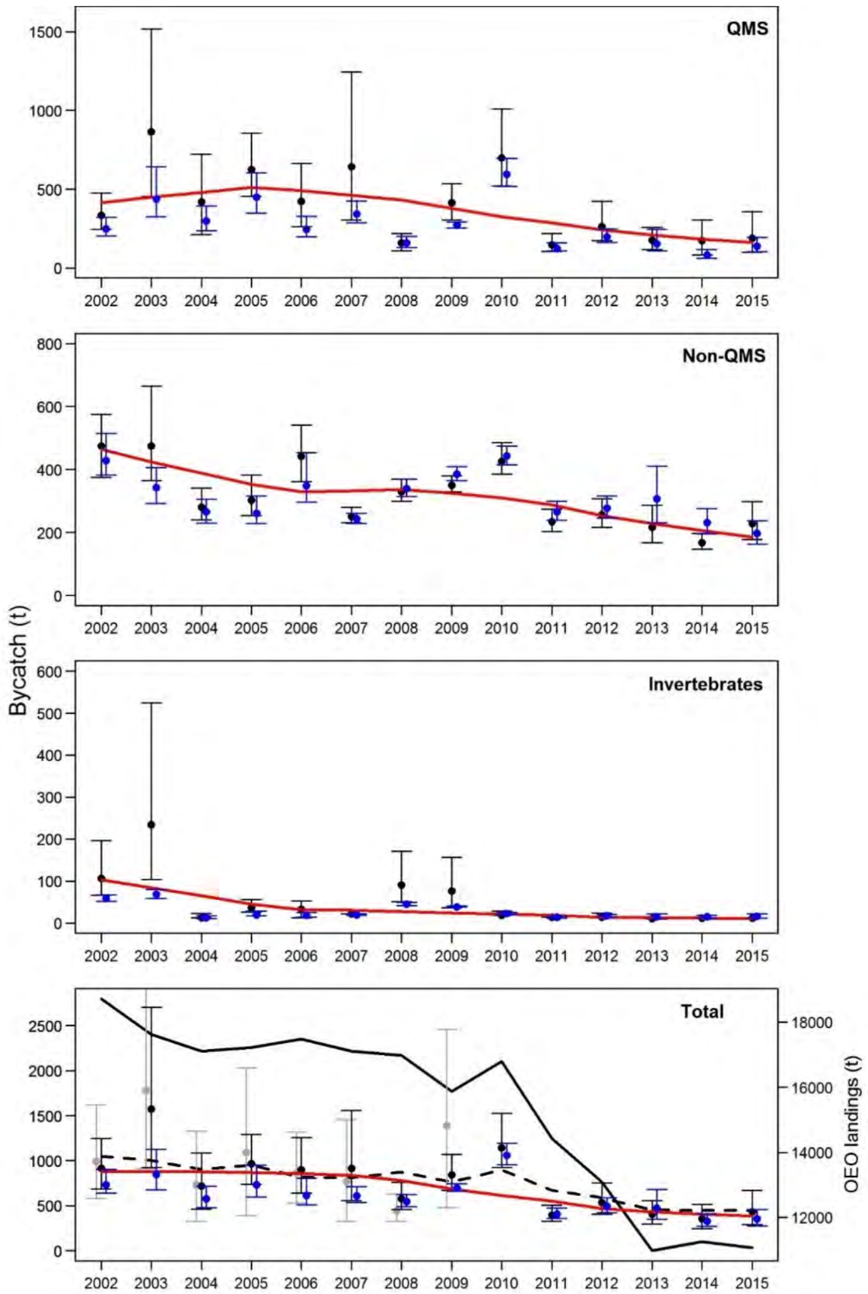

Figure 9.43: Annual estimates of bycatch in the oreo target trawl fishery for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method. Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence 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.

TRENDS IN BYCATCH BY SPECIES FROM THE OREO TRAWL FISHERY

Anderson (2017) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2013–14. The following conclusions were made:

- The most commonly caught bycatch species were orange roughy (*Hoplostethus atlanticus*, ORH), unspecified shark (SHA), and hoki (HOK).
- Of the 120 bycatch species examined, 2 showed a significant decrease in catch over time and 10 showed a significant increase in catch.

- The species showing the greatest decline were dark ghost shark (GSH), unspecified shark (SHA),[3] and ling (LIN) (Figure 9.44).
- The species showing the greatest increase were pale ghost shark (GSP), Baxter's lantern dogfish (*Etmopterus baxteri*, ETB), and ridge-scaled rattail (*Macrourus carinatus*, MCA) (Figure 9.44).

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 9.45). Discards mainly comprised non-QMS species, but included a varying amount of the target species in most years.

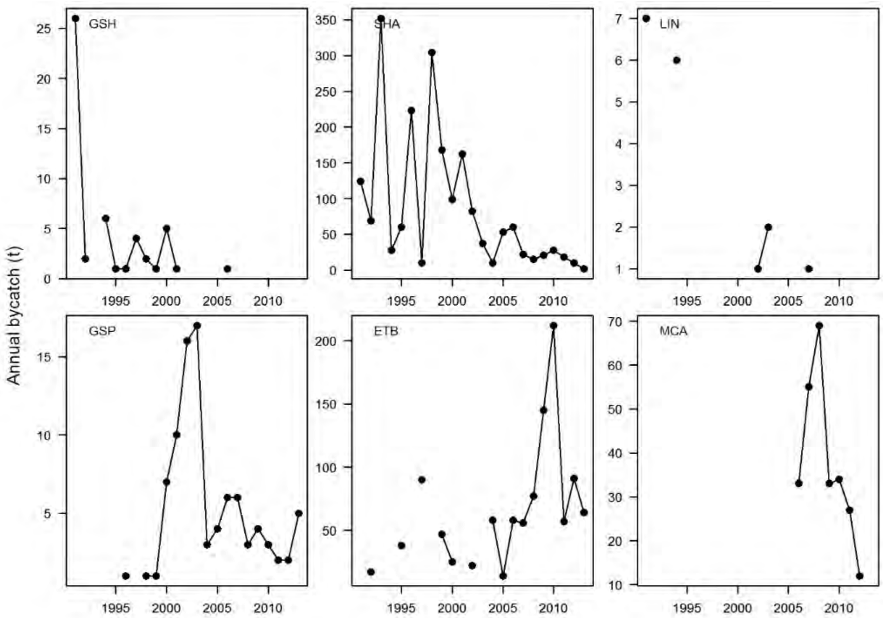

Figure 9.44: Annual bycatch estimates in the oreo trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2013–14. See text above for explanation of the species codes. Sharks (SHA) may have been identified to an increasingly higher 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.

[3] Notably SHA is a generic code whose decline is probably due to better species level identification of sharks over time.

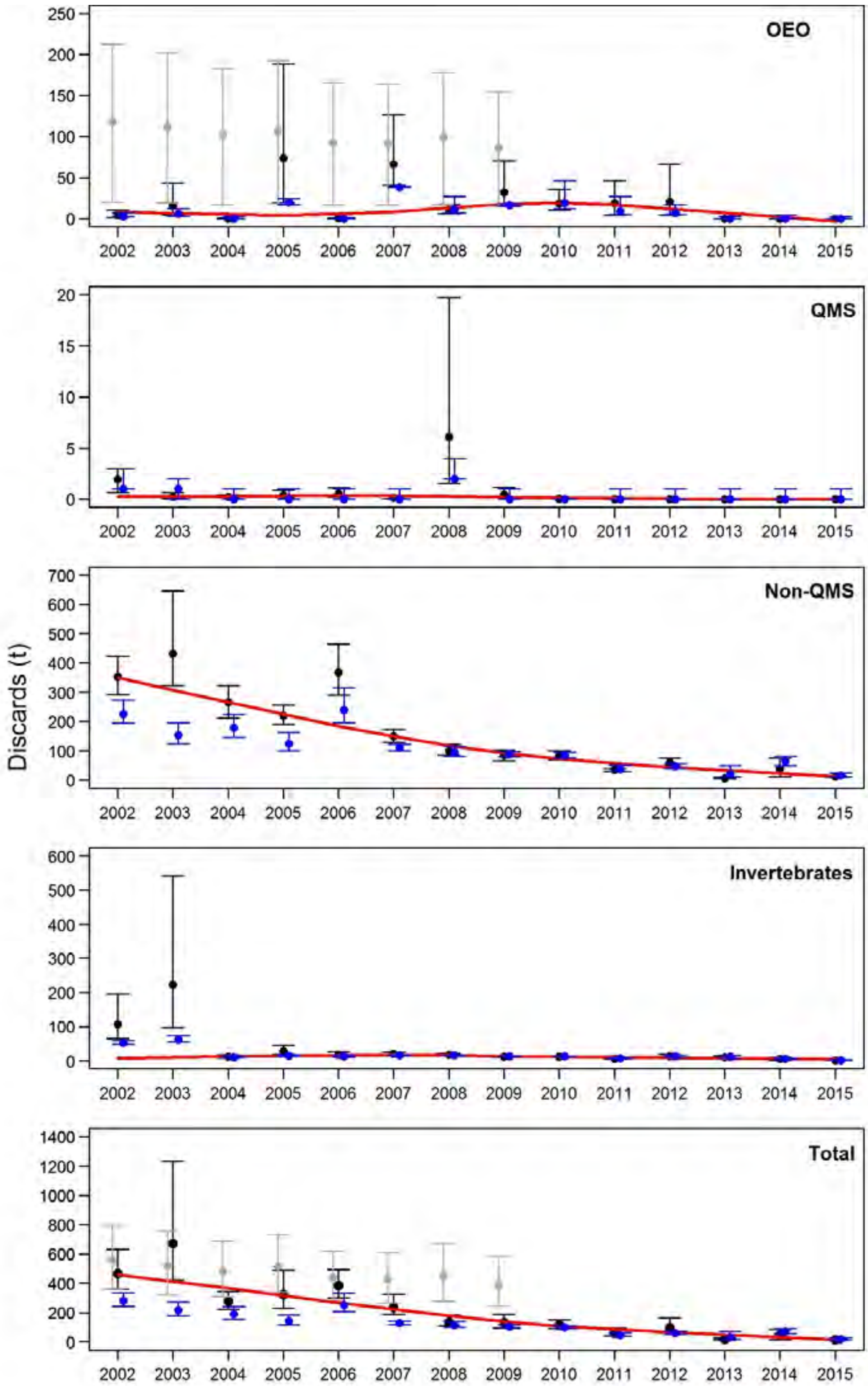

Figure 9.45: Annual estimates of discards in the oreo fishery, for species categories for 2001–02 to 2014–15. Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

1.3.18 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 based, with effort based on the number of trawls (Anderson & Edwards, in prep).

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 deepwater fisheries assessed. The long-term level of observer coverage in the orange roughy, oreo, arrow squid, southern blue whiting 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 trawls in the fishery ranged from 142 to 535, but was over 300 trawls in most years (2.7 to 15.5% of the total number of trawls). The number of vessels observed in each year ranged from 3 to 8 (equivalent to 33–75% of the fleet) and was relatively constant – 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.

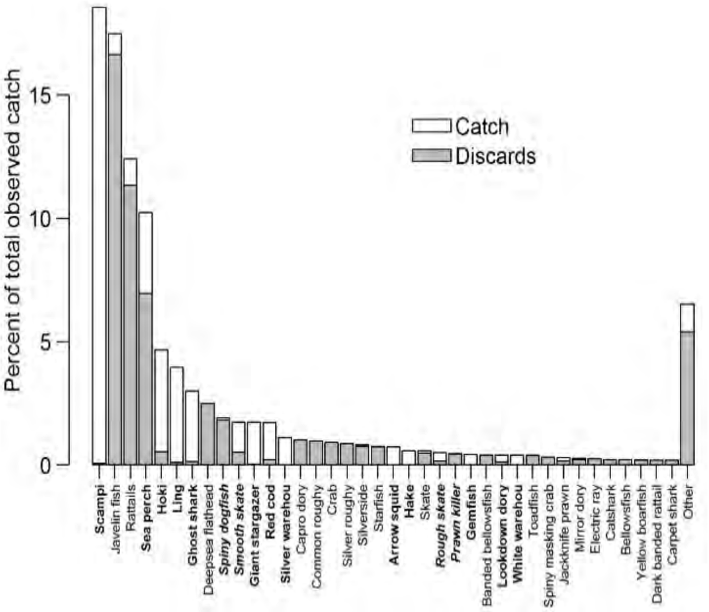

Figure 9.46: Percentage of the total catch contributed by the main bycatch species (those representing 0.2% or more of the total catch) in the observed portion of the scampi fishery, 2001–02 to 2015–16, and the percentage discarded. The 'Other' category is the sum of all other bycatch species (fish and 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.

TRENDS IN BYCATCH BY SPECIES FROM THE SCAMPI TRAWL FISHERY

Anderson & Edwards (in prep) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 2002–03 to 2015–16. The following conclusions were made:

- The most commonly caught bycatch species were javelinfish (*Lepidorhynchus denticulatus*, JAV), unspecified rattails (Macrouridae, RAT), and sea perch (*Helicolenus* spp., SPE) (Figure 9.48).

- Of the 22 bycatch species examined, 5 showed a significant decrease in catch over time while none showed an increase in catch.
- The species showing significant declines were ling (*Genypterus blacodes* LIN), gemfish (*Rexea* spp. SKI) silver warehou (*Seriolella punctata*, SWA), hake (*Merluccius australis*, HAK), and toadfish (*Neophrynichthys* sp. TOA).

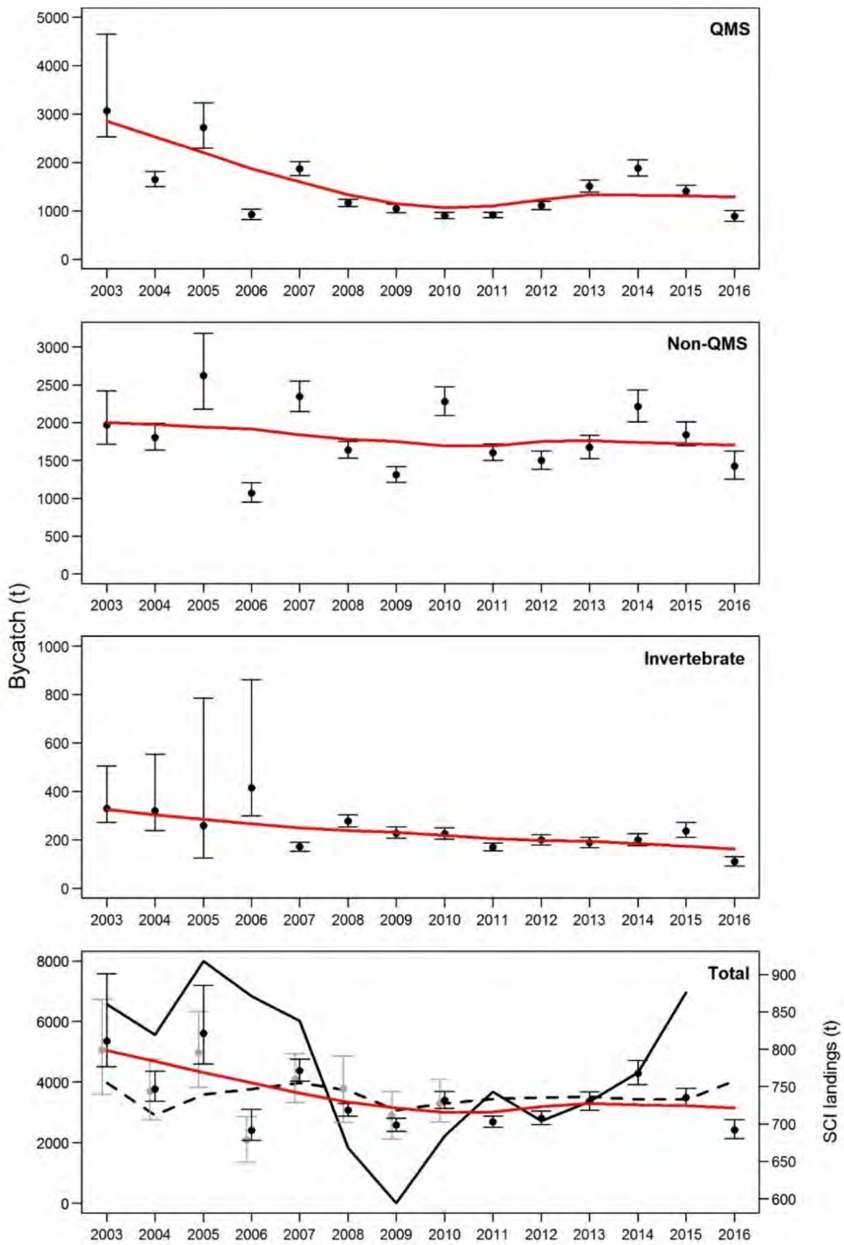

Figure 9.47: Annual estimates of bycatch in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16. Also shown (in grey) are estimates of Total bycatch calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. 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).

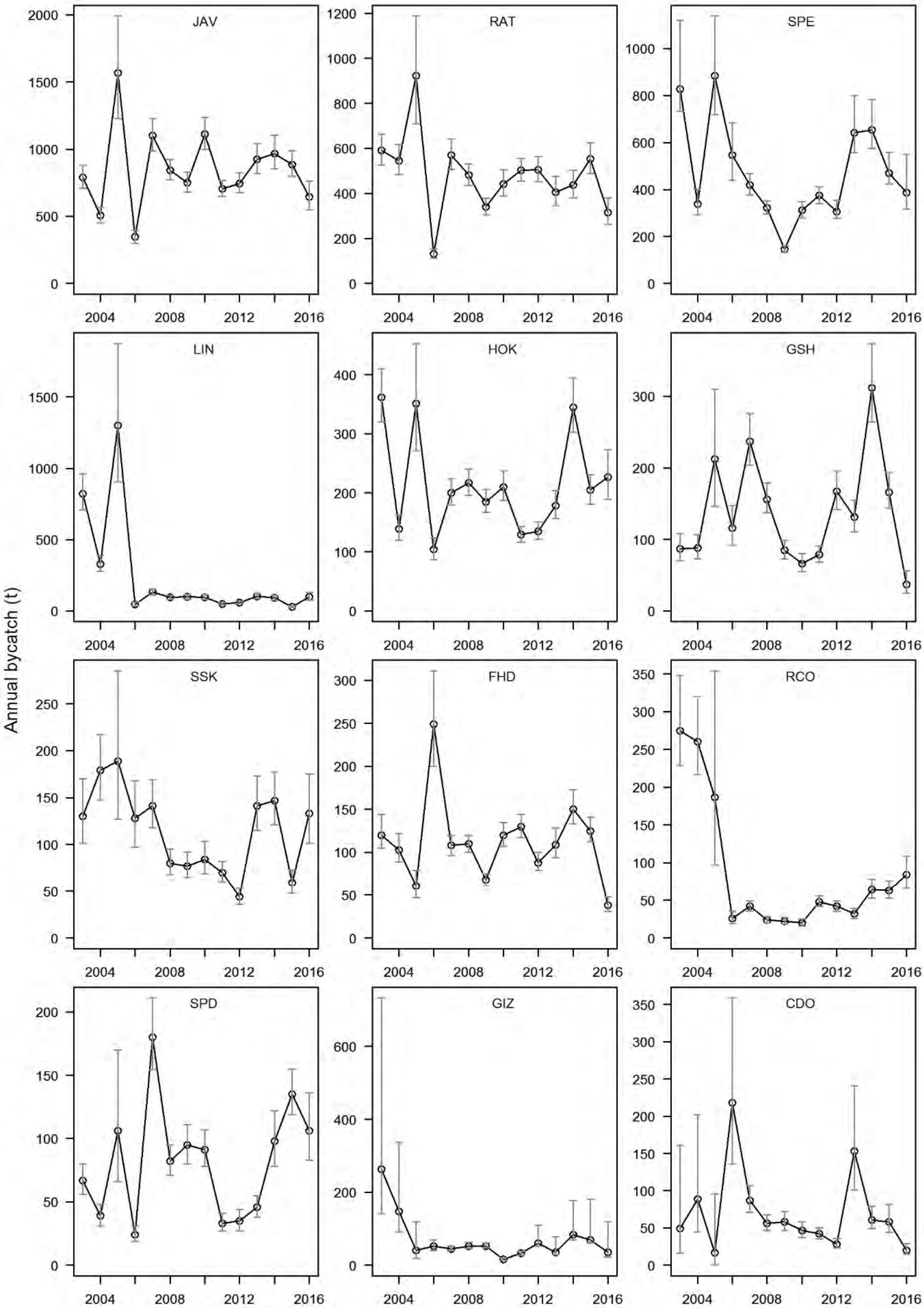

Figure 9.48: Annual bycatch estimates in the target scampi trawl fishery for the species which have the most bycatch between 2002–03 and 2015–16, with 95% c.i.s, in descending order of total catch. See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots.

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 9.49). 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.

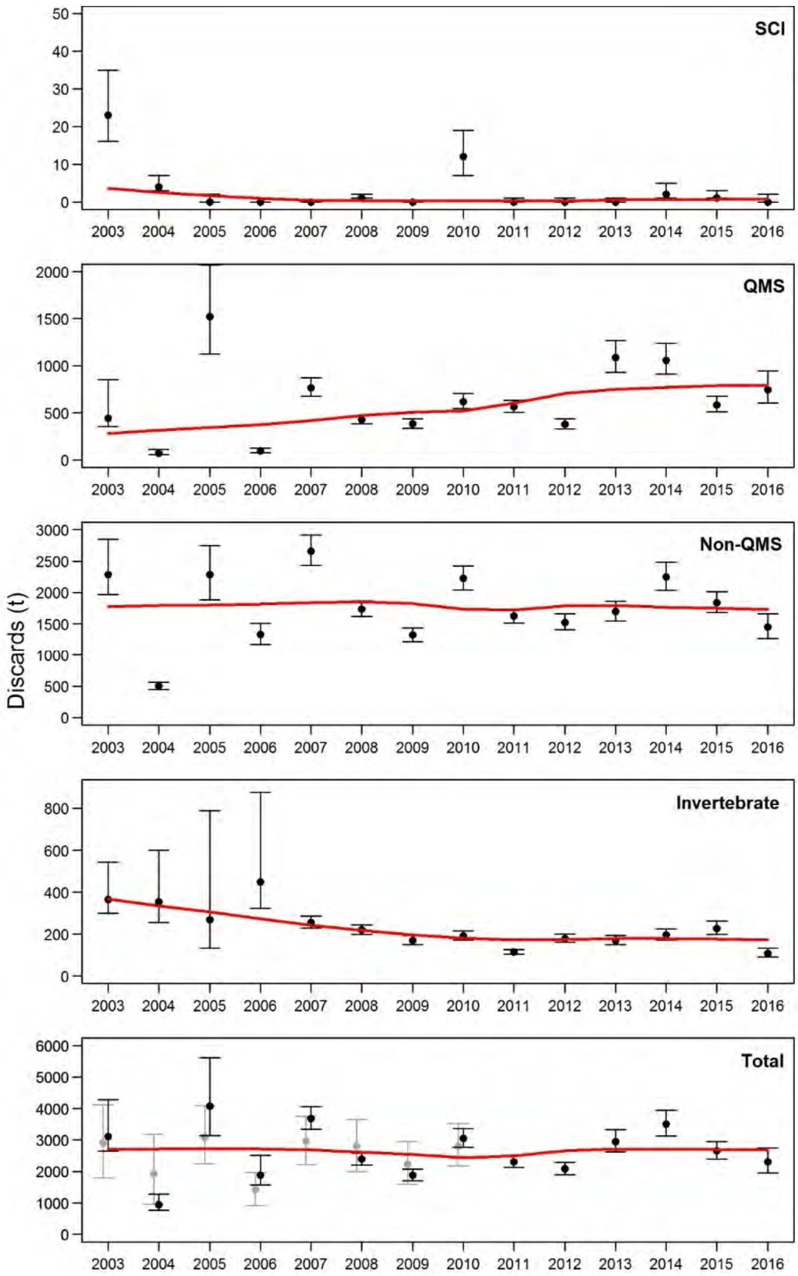

Figure 9.49: Annual estimates of discards in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16. Also shown (in grey) are estimates of Total discards calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted regression to annual discards.

1.3.19 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 ratio estimator used in these analyses to calculate bycatch and discard rates was based on the number of hooks set. The ratios were applied to hook number totals calculated from commercial catch-effort data to make annual estimates for the target fishery as a whole.

Linear mixed-effect models (LMEs) were used to identify key factors influencing variability in the observed rates of bycatch and discarding, resulting in an area based stratification for the calculations.

Between 1992–93 and 2011–12 only 9% of the vessels operating in this fishery were observed (24 vessels) but these tended to be the main operators (including most of the larger autoliners) and accounted for up to 52% (by catch) and 60% (by effort) of the total fishery, although coverage was low (less than 10%) prior to 1999–2000).

Ling made up 68% of the total estimated catch from all observed sets targeting ling between 1992–93 and 2011–12, and spiny dogfish (much of which was discarded[4]) about a further 13% (Figure 9.50). About half of the remaining 19% of the catch comprised other commercial species; especially ribaldo (*Mora moro*) (2.9%), smooth skates (*Dipturus innominatus*) (1.8%), red cod (*Pseudophycis bachus*) (1.7%), rough skates (*Zearaja nasuta*, 1.7%), and sea perch (*Helicolenus* spp.) (1.2%). Altogether, 93% of the observed catch was comprised of QMS species. Over 230 species or species groups were identified by observers, the majority 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*). A large weight of echinoderms, especially starfish (of which about 39 t were observed caught during the period), anemones, crustaceans, and other invertebrates were also recorded by observers.

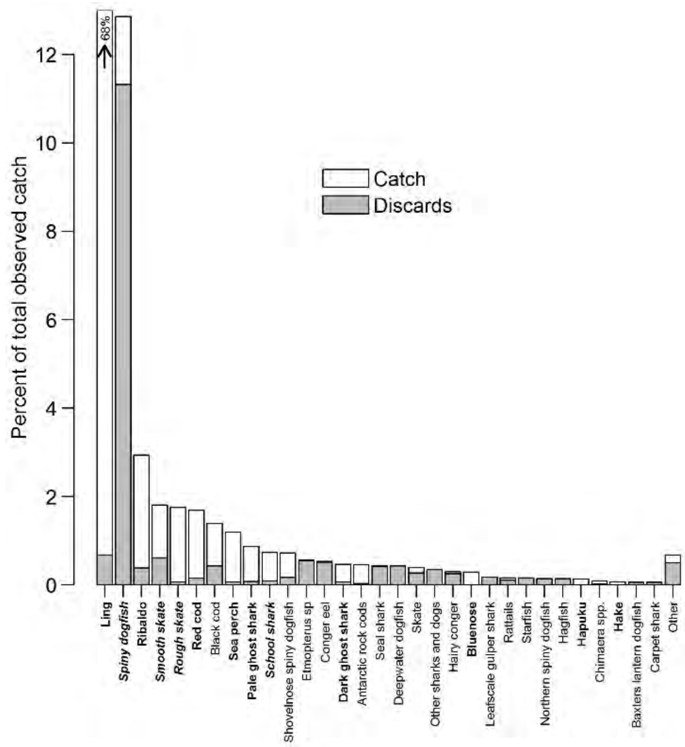

Figure 9.50: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch) in the observed portion of the ling longline fishery, 1992–93 to 2011–12 and the percentage discarded. QMS species are shown in bold.

Total annual bycatch estimates for 1992–93 to 2011–12 were 2100–3900 t, compared with approximate target species catches in the same period of 3500–9800 t. Bycatch weights decreased slowly during the period, in line with decreasing effort in the fishery (Figure 9.51).

TRENDS IN BYCATCH BY SPECIES FROM THE LING BOTTOM-LONGLINE FISHERY

Ballara (2015) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1992–93 to 2012–13. The following conclusions were made:

- The most commonly caught bycatch species were spiny dogfish (SPD), ribaldo (*Mora moro*, RIB), and smooth skate (*Dipturus innominatus*, SSK).

[4] Spiny dogfish can legally be discarded under Section 6 of the Fisheries Act 1996.

- Of the 104 bycatch species examined, 5 have shown a decrease in catch over time and 9 an increase in catch.
- Among the species showing the greatest decline were bluenose (*Hyperoglyphe antarctica*, BNS), Ray's bream (*Brama brama*, RBM), and hapuku (*Polyprion oxygeneios*, HAP) (Figure 9.52).
- The species showing the greatest increase were the hairy conger (HCO), hagfish (*Eptratretus cirrhatus*, HAG), and pale ghost shark (*Hydrolagus bemisi*, GSP) (Figure 9.52).

Total annual discard estimates for 1992–93 to 2011–12 were 1200–2500 t, and generally decreased during the period (Figure 9.53). About 45–70% of these discarded fish were quota species, mainly spiny dogfish,[5] the remainder being non-quota, generally non-commercial, species. Ling were discarded in small amounts (40–250 t per year), these generally being attributable to fish being lost on retrieval or predated by marine mammals and birds.

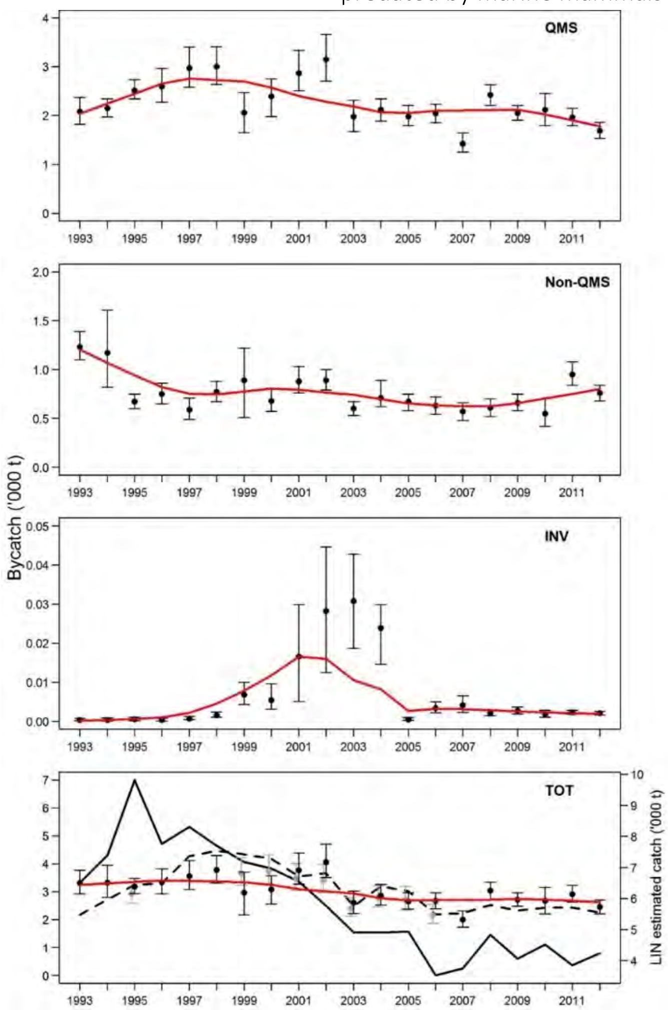

Figure 9.51: Annual estimates of bycatch in the ling longline fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 1992–93 to 2011–12. Also shown (in grey) are earlier estimates of total bycatch calculated for 1994–95 and 1998–99 to 2005–06 (Anderson 2008). Error bars indicate 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 estimated commercial longline-catch of ling, and the dashed line shows annual effort (number of hooks), scaled to have mean equal to that of total bycatch.

[5] Spiny dogfish can legally be discarded under Schedule 6 of the Fisheries Act 1996.

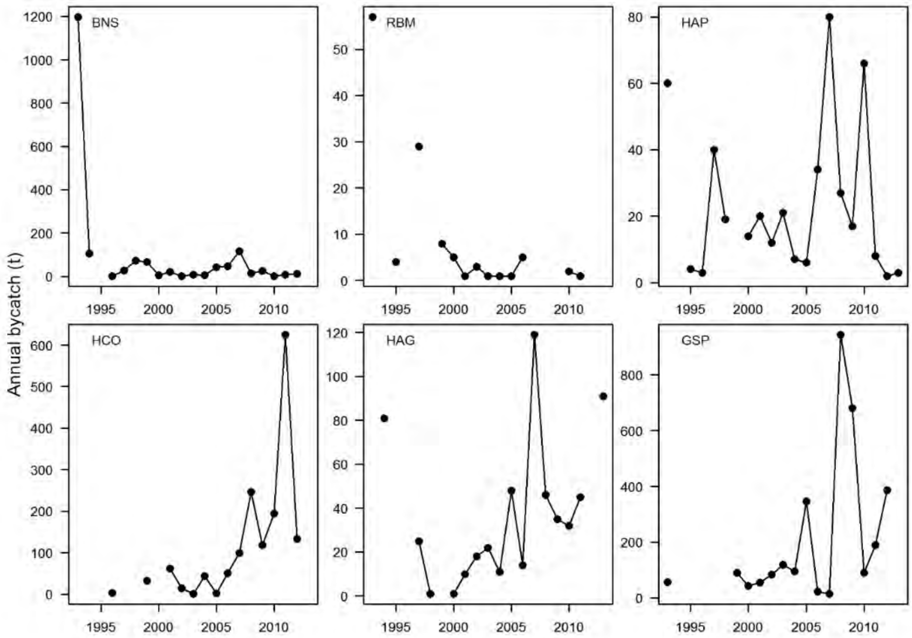

Figure 9.52: Annual bycatch estimates in the ling longline fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1992–93 and 2012–13. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3.1). See text above for species codes.

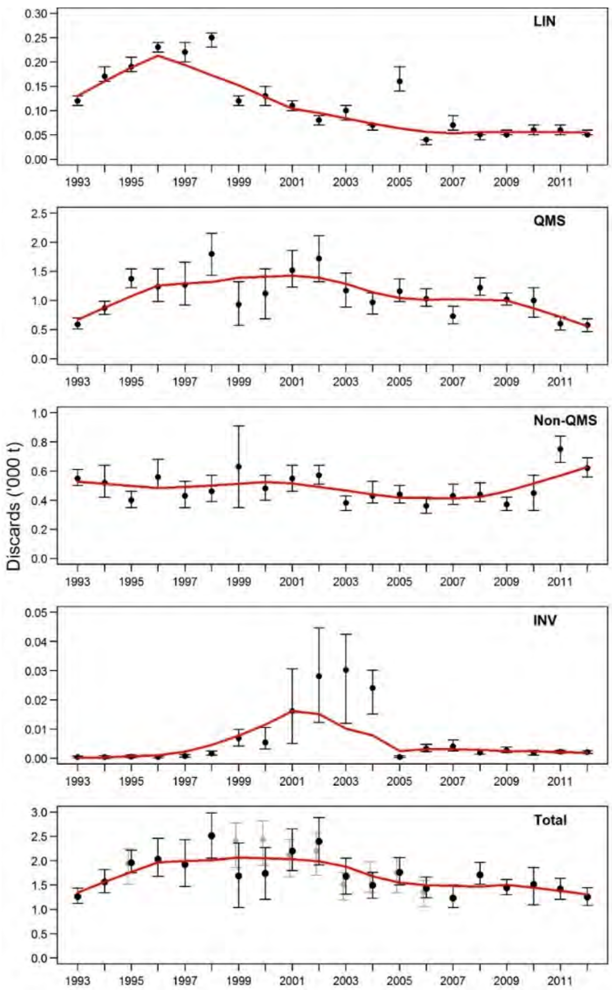

Figure 9.53: Annual estimates of discards in the ling longline fishery, for ling (LIN), QMS species, non-QMS species, invertebrates (INV), and overall for 1992–93 to 2011–12. Also shown (in grey) are earlier estimates of total discards calculated for 1994–95 and 1998–1999 to 2005–06 (Anderson 2008). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

1.3.20 TUNA LONGLINE FISHERY

The New Zealand tuna longline fishery was dominated by the foreign licensed vessels during the 1980s, then was comprised of 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).

The Japanese charter fleet mainly targeted southern bluefin tuna off the west coast South Island (WCSI), and domestic vessels targeted southern bluefin tuna and bigeye tuna and the fishery was concentrated on the east coast of the North Island (ECNI) with some fishing for southern bluefin tuna on 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 is in prep (Griggs et al., in prep) for 2010–11 to 2014–15.

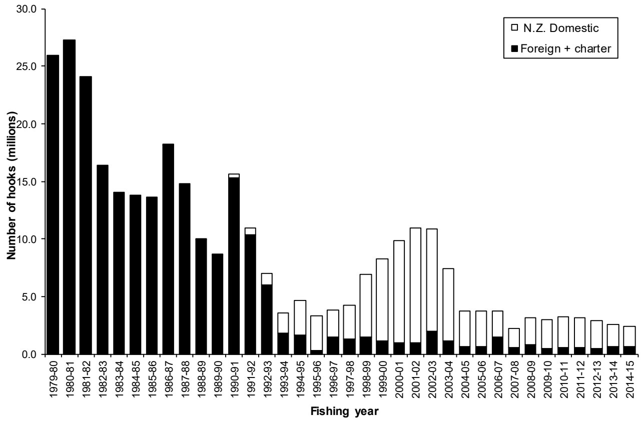

Figure 9.54: Effort (hooks set) in the tuna longline fishery. Black bars are Foreign and Charter vessels, white bars are NZ domestic vessels.

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, Ray’s bream, and albacore tuna, 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 non-target species were albacore, porbeagle shark, lancetfish, dealfish, deepwater 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.

Table 9.2: Species composition of observed tuna longline catches. Number of fish observed are shown for 2006–07 to 2009–10 and all fish observed since 1988–89. Top 30 species.

	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
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

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 has been banned in New Zealand waters. Most sharks were discarded in 2014–15, except for some mako and school shark retained for their flesh.

Most albacore, swordfish, yellowfin tuna, moonfish and Ray's bream were retained. Most bigscale pomfret, and rudderfish were discarded, while butterfly tuna, escolar and oilfish were often retained, with some year and fleet differences. Almost all deepwater dogfish, dealfish and lancetfish were discarded.

1.3.21 ALBACORE TUNA TROLL FISHERY

This fishery was carried out by small domestic vessels fishing over the summer months mainly on the west coast of the North 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 was 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 as 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

1.3.22 SKIPJACK TUNA PURSE SEINE FISHERY

Skipjack tuna was 97.0 % of the catch observed on purse seine vessels in New Zealand waters in 2015 and 2016. Catch composition from four observed purse seine trips operating within New Zealand fisheries waters in 2015 and 2016 can be seen in Table 9.4.

Table 9.4: Catch composition from six observed purse seine trips operating within New Zealand fisheries waters in 2015 and 2016. [Continued on next page]

Common name	Scientific name	Observed catch weight (kg)	% of catch
Skipjack tuna	*Katsuwonus pelamis*	3 478 271	97.05
Jack mackerel	*Trachurus* spp.	80 573	2.25
Sunfish	*Mola mola*	8 867	0.25
Blue mackerel	*Scomber australasicus*	5 646	0.16
Frigate tuna	*Auxis thazard*	2 839	0.08
Spine-tailed devil ray	*Mobula japanica*	1 641	0.05
Striped marlin	*Tetrapturus audax*	1 190	0.03
Jack mackerel	*Trachurus novaezelandiae*	1 030	0.03
Albacore tuna	*Thunnus alalunga*	734	0.02
Blue marlin	*Makaira mazara*	650	0.02
Marlin unspecified		600	0.02
Frostfish	*Lepidopus caudatus*	390	0.01
Mako shark	*Isurus oxyrinchus*	385	0.01
Jellyfish		266	0.01
Slender tuna	*Allothunnus fallai*	177	<0.01
Southern bluefin tuna	*Thunnus maccoyii*	130	<0.01
Flying fish	Exocoetidae	92	<0.01
Bigeye thresher shark	*Alopias superciliosus*	80	<0.01
Yellowfin tuna	*Thunnus albacares*	80	<0.01
Squid	Teuthoidea	56	<0.01
Giant stargazer	*Kathetostoma giganteum*	50	<0.01
Stingray	Dasyatididae	45	<0.01
Smooth skate	*Dipturus innominatus*	35	<0.01
Discfish	*Diretmus argenteus*	30	<0.01

Table 9.4 [Continued]:

Common name	Scientific name	Observed catch weight (kg)	% of catch
Porcupine fish	*Allomycterus jaculiferus*	18	<0.01
Dolphinfish	*Coryphaena hippurus*	15	<0.01
Octopus		12	<0.01
Ray's bream	*Brama brama*	10	<0.01
Snapper	*Pagrus auratus*	10	<0.01
School shark	*Galeorhinus galeus*	8	<0.01
Electric ray	*Torpedo fairchildi*	6	<0.01
Pilotfish	*Naucrates ductor*	6	<0.01
Barracouta	*Thyrsites atun*	5	<0.01
Pelagic ray	*Pteroplatytrygon violacea*	5	<0.01
Ling	*Genypterus blacodes*	5	<0.01
Tarakihi	*Nemadactylus macropterus*	4	<0.01
Flatfish		3	<0.01
John Dory	*Zeus faber*	3	<0.01
Kingfish	*Seriola lalandi*	3	<0.01
Skate		3	<0.01
Dealfish	*Trachipterus trachypterus*	2	<0.01
Pale ghost shark	*Hydrolagus bemisi*	1	<0.01
Spotted gurnard	*Pterygotrigla picta*	1	<0.01
Leatherjacket	*Parika scaber*	1	<0.01
Louvar	*Luvaris imperialis*	1	<0.01
NZ northern arrow squid	*Nototodarus gouldi*	1	<0.01
Opah	*Lampris immaculatus*	1	<0.01
Starfish		1	<0.01
Unidentified		1	<0.01

1.4 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).

Comparison of estimates of total bycatch over time from all the deepwater trawl fisheries (Figure 9.55) shows the substantial contribution from the large hoki/hake/ling fisheries (2016–17 hoki total TACC of 150 000 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 (2016–17 scampi total TACC of 1244 t) and the arrow squid fishery (2016–17 arrow squid total TACC of 82 120 t).

Some general trends were identified in some fisheries, especially those examined in recent MPI projects where the determination of trends in the rates and levels of bycatch over time was an explicit objective (Table 9.6).

Table 9.5: Utilisation rates. Kilograms of discards per kilogram of target species catch. The numbers are the most recent estimate (mean of the most recent four years available) from referenced reports.

Fishery	Discards/target species catch (kg)
Arrow squid trawl	0.12
Ling longline	0.34
Hoki/hake/ling trawl	0.05
Jack mackerel trawl	0.01
Southern blue whiting trawl	0.01
Orange roughy trawl	0.04
Oreo trawl	0.01
Scampi trawl	3.83

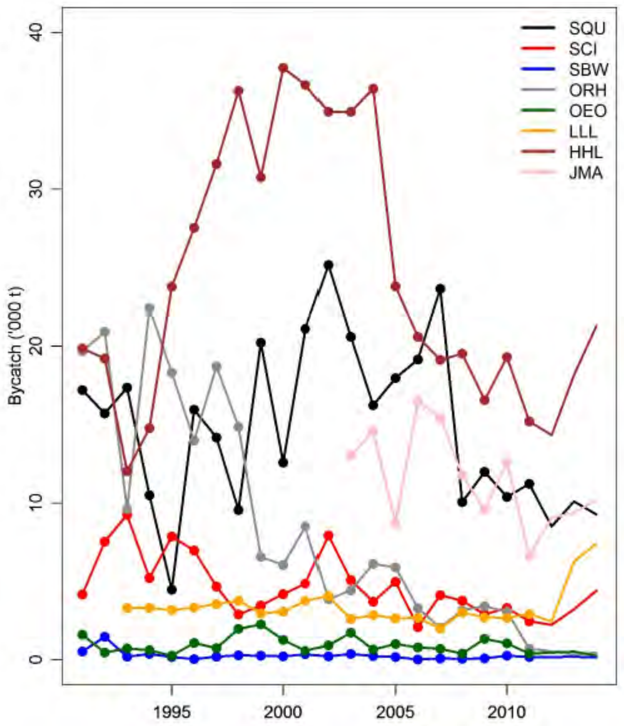

Figure 9.55: Comparison of total estimated bycatch for all the deepwater trawl fisheries 1990–91 to 2013–14. Dots are precision based estimates; no dots are coarse based estimates (see Anderson 2017 for more details). For species codes see Table 9.1).

Table 9.6: Trends in non-protected species bycatch from recent MPI projects where trend determination was an objective.

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 decresing 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.
Jack mackerel trawl	Linear regressions showed decreasing bycatch of QMS and non-QMS species, and increasing bycatch of invertebrate species and spiny dogfish, but these trends were not statistically significant. Linear regressions showed decreasing discards of non-QMS species, invertebrate species and spiny dogfish, and increasing discards of jack mackerels and QMS species, but these trends were only significant for jack mackerel.
Orange roughy trawl	Increased non-commercial 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.
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.
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).
Ling longline	Linear regression modelling of observer catch data indicated increasing bycatch rates (kg/hook) for both QMS species and non-QMS species in LIN 2, and for QMS species in COOK; and decreasing bycatch rates for both QMS species and non-QMS species in BNTY, and QMS species in LIN 4. There were also decreasing discard rates of ling and QMS species in some areas, notably the Bounty and Campbell Plateaus, and increasing discard rates for both QMS species and non-QMS species in LIN 2.

Anderson (2017) analysed temporal (1990–91 to 2013–14) bycatch trends for individual species or species groups for seven deepwater trawl and one bottom-longline (ling) fisheries. A summary of the bycatch regression slope coefficients for each species and fishery is provided in graphical form in Appendix 19.10.1. This showed a consistent increase (in six or more of the eight fisheries) for deepsea skates (*Notoraja* spp.), Baxters lantern dogfish (*Etmopterus baxteri*), pale ghost shark (*Hydrolagus bemisi*), and javelinfish (*Lepidorhynchus denticulatus*); and consistent decline for bluenose (*Hyperoglyphe antarctica*), dark ghost shark (*Hydrolagus novaezealandiae*), and skates (*Rajidae* and *Arhynchobatidae*). 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.

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10 CHONDRICHTHYANS (SHARKS, RAYS AND CHIMAERAS)

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	All of the New Zealand EEZ and Territorial Sea.
Focal localities	This differs depending upon the species or fishery examined.
Key issues	Sustainability of fisheries extractions.
Emerging issues	Risk assessment of fisheries extractions.
MPI research (current)	POS2013-01 *Abundance indicators for blue, porbeagle and mako sharks*; SEA2014-03 *Shark fin and dressed weight conversion factors for selected sharks*; SEA2013-16 *Shark, ray and chimaera qualitative risk assessments*; SEA2013-10 *Porbeagle shark movements from tagging*; HMS2013-02 *Catch composition of porbeagle sharks in tuna longline fisheries*; HMS2014-02 *Catch composition of mako sharks in tuna longline fisheries*; HMS2014-05 *Stable isotope analysis of highly migratory species*; SEA2013-11 *Trends in blue shark catch rates in tuna longline fisheries*; ZBD2011-01 *Diets of highly migratory species*; ZBD2011-01 *Factors affecting the distribution of highly migratory species*; HHS2013-01 *Hammerhead shark movements estimated by tagging*; INS2014-01 *Abundance indicators for 8 sharks and chimaeras*; ENV2014-02 *Age and growth of 7 sharks and rays*; SPO2015-01 *Characterisation and CPUE of rig stocks.*
Other government research (current)	DOC CSP research: MIT2013-04 *Basking shark mitigation: detection and avoidance*; MIT2011-01 *Protected rays – mitigate captures and assess survival of live-released animals*; DOC14304 *Review of fishery interactions and biology of oceanic whitetip shark.* MBIE project: C01X0905 *Conservation of New Zealand's threatened iconic marine megafauna*.
University research	Biological and behavioral research is currently being carried out at Waikato, Otago, Victoria and Auckland universities on a variety of species (including white shark, Lucifer's dogfish, longnose spookfish, Pacific spookfish, prickly dogfish, broadnose sevengill shark, blue shark) and on mitigating the bycatch of sharks on longlines. This list may not include all university research.
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 (Ministry for Primary Industries 2015).

Note: This chapter has not been revised since the AEBAR 2015.

10.1 CONTEXT

Chondrichthyans (cartilaginous fishes) comprise all fish species (except lampreys and hagfish) that lack true bone in their skeletons, 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 prohibited as target species. 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 (southern bluefin tuna), Commission for the Conservation of Antarctic Marine Living

Resources (toothfish), and the South Pacific Regional Fisheries Management Organisation (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 in June 1999 and subsequently adopted by the November 1999 FAO Conference. The overarching goal of the IPOA is: 'to ensure the conservation and management of sharks and their longterm 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 that came into effect in October 2008 (Ministry of Fisheries 2008), this has since been superseded by the NPOA-Sharks 2013 (Ministry for Primary Industries 2013). 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.*'

It aims to achieve this purpose by identifying 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;
- Domestic engagement and partnerships;
- Non-fishing threats;
- International engagement;
- Research and information.

It is a comprehensive plan to focus efforts to improve our research and information on shark populations, and base conservation and management actions on an assessment of risks.

The NPOA-Sharks applies to all chondrichthyans that are found within New Zealand's Exclusive Economic Zone (EEZ) and Territorial Sea (New Zealand fisheries waters), 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 10.1 provides a list of all 117 known New Zealand chondrichthyans, along with their management class and IUCN and Department of Conservation threat classes.

10.2 BIOLOGY

The population dynamics of chondrichthyans differ markedly from those of bony fishes. Chondrichthyans have a mammal-like reproductive strategy of producing 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., blue shark 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, mako shark; Mollet et al. 2000, Walker 2005). Fecundity may increase with the size of females (e.g., rig and school shark; Francis & Mace 1980, Walker 2005) so if human activities reduce the average size of females in a

[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.

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 potential for density-dependent compensatory mechanisms that might boost reproductive output at low population sizes.

Many cartilaginous fishes are also long-lived and slow growing, further reducing their capacity for recovering from population declines. Many species have ages at maturity greater than 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, Pikitch et al. 2008, Simpfendorfer & Kyne 2009).

Six feeding studies have been carried out in the last few years on a suite of middle depth to deepwater chondrichthyans, mainly using stomach content data collected during Chatham Rise trawl surveys (Jones 2008, 2009, Dunn et al. 2010a, 2010b, 2013, Forman & Dunn 2012). The diets of blue, porbeagle and mako sharks have been analysed using samples collected by observers on tuna longline vessels (Horn et al. 2013). Fish and squid were the primary prey of shark species, with chimaeras having a diet dominated by benthic invertebrates, and skates also feeding on benthic and natant invertebrates. There was evidence of both depth- and diet-related niche separation. In one study, DNA testing was used to identify stomach contents. The importance of discards in the diet of some sharks and rays was highlighted. In a seventh study, juvenile rig were found to feed mainly on benthic crustaceans such as mud crabs and snapping shrimps in estuaries around New Zealand (Getzlaff 2012).

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. However the rapid increase in demand for, and value of, shark fins over the last two decades has resulted in a rapid increase in chondrichthyan fishing mortality throughout the world, although there is evidence of a sharp decline in the shark fin trade since 2011 (Dent & Clarke 2015), and 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. (Live shark finning is an offence under the Animal Welfare Act 1999). Shark finning was banned in New Zealand in October 2014. The results of this ban are expected to become apparent in the next fishing year.

Chondrichthyans are caught by most fishing methods, although trawling, netting and lining are the most important. Chondrichthyans are caught in nearly all parts of the world, ranging from tropical to arctic/antarctic waters, and from estuaries and shallow coastal waters to the deepest areas fished. Historically, most chondrichthyan catches worldwide have been taken as bycatch in fisheries for other target species. However, the increased value of shark fins has driven a move towards target fishing for some shark species elsewhere in the world, and increased utilisation of incidentally caught sharks. Consequently reported global landings of chondrichthyans increased steadily up to almost 900 000 t in the early 2000s but have been declining since then (Worm et al. 2013). 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). Furthermore, the survival of discarded chondrichthyans has rarely been quantified: measures of mortality rates of chondrichthyans at the time they are hauled to a fishing vessel are available for some species (Francis et al. 1999a, Campana et al. 2009, Griggs & Baird 2013), but estimates of subsequent survival of live releases are rare (Moyes et al. 2006, Campana et al. 2009, Musyl et al. 2011, Hutchinson et al. 2013).

Despite these uncertainties, there is ample evidence that 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 Organisations are paying greater attention to the need to manage species that occur in international waters or straddle the national waters of multiple countries. 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. However it is not clear that this requirement has been effective in reducing catches (Clarke et al. 2012, Worm et al. 2013).

10.4 STATE OF KNOWLEDGE OF FISHERIES INTERACTIONS IN NEW ZEALAND

A total of 117 chondrichthyans are known from New Zealand waters (including the Ross Sea), however that number is expected to grow slightly as taxonomic studies continue on deepwater species. Of these species, 12 are chimaeras, 30 are skates and rays, and 75 are sharks. Many New Zealand species also occur elsewhere in the world (some have worldwide distributions) but a high percentage (30%) are endemic to New Zealand. New Zealand's chondrichthyan fauna is small compared with that in Australia, which has more than 322 species (Last & Stevens 2009), but that partly reflects New Zealand's lack of tropical environments. The high percentage of endemic species makes New Zealand's fauna unique and distinctive.

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, and shark ecotourism (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 (Ministry for Primary Industries 2013, 2015), seven are fully protected (Francis & Lyon 2012), two cannot be targeted, and the remainder are Non-QMS species (Appendix 10.1). Due to reporting requirements commercial landings of chondrichthyans are relatively well known, but less is known about recreational and customary catches.

A nationwide survey from 1 October 2011 to 30 September 2012 provides the most reliable estimates of recreational chondrichthyan catches (Table 10.1) (Wynne-Jones et al. 2014). The majority of the recreational catch is from inshore QMS species; mako is the only shark listed that is not normally considered an inshore species. '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/bycatch in the gamefish charter boat fishery, so estimates for mako are potentially underestimates as the survey was not designed to sample gamefishers on charter boats. Estimates in tonnes are only available for rig and spiny dogfish and these constitute 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 fishery.

Commercial catches of chondrichthyan species during the five-year period 2009−10 to 2013−14 are shown in Table 10.2 and Figure 10.1. Spiny dogfish produced by far the greatest catches, followed by school shark. Dark ghost shark, rough skate, rig and elephantfish formed a second tier of species, and blue shark, pale ghost shark, smooth skate, carpet shark and seal shark formed a third tier; the remaining species had relatively low catches (less than 215 t per year on average). Unspecified sharks and unspecified dogfish were both important categories, indicating that fishers were not accurately recording all catches to species level. Reported discards in 2012–13 to 2013–14 were significant for spiny dogfish, seal shark, carpet shark, shovelnose dogfish and other deepwater and unspecified sharks (Table 10.3). Live releases of seven specified chondrichthyans are permitted under Schedule 6 of the Fisheries Act, and from 2006−07 such releases were not counted against quota (Table 10.4). Spiny dogfish may also be discarded dead, but are be counted against a fisher's Annual Catch Entitlement (ACE) and the total allowable catch limit for that species against quota. Live releases in 2012–13 to 2013–14 were significant for blue, porbeagle and mako sharks, and smooth skates (Table 10.3). The conditions of Schedule 6 releases have been amended for mako, porbeagle, and blue shark. From 1 October 2014, fishers were 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 Annual Catch Entitlement (ACE) and the total allowable catch limit for that species. The survival rate of discarded and released sharks is unknown, and probably varies enormously with species, fishing method, handling, and other factors.

10.4.1 QMS SPECIES

The eleven chondrichthyans managed under the QMS are shown in Table 10.4 with their Total Allowable Commercial Catches (TACCs) and 2013−14 landings. Landings of all but two species (rough skate and elephantfish) were below the TACCs.

QMS chondrichthyans are treated in detail in MPI's annual Fisheries Assessment Plenary reports (Ministry for Primary Industries 2014, 2015) 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 only the assessment for SPO 7 was accepted and adopted by the MPI Southern Inshore Working Group. The status of other stocks has been estimated from trends in standardised CPUE and trawl surveys.

A summary of the status of the stocks of QMS chondrichthyans is given in Appendix 10.2. Stock status has been estimated for six of the 11 QMS chondrichthyans, and 18 of the 45 stocks. None of the stocks was considered to be below the 'hard limit' reference point, one stock (SPO 7) was considered unlikely (<40%) to be below the 'soft limit'. No stocks were considered to be in an 'overfishing' state; the remainder of the stocks were considered to be in a favourable state.

10.4.2 PROTECTED SPECIES

Seven chondrichthyans are currently protected in New Zealand fisheries waters: white shark (also known as white pointer shark) was protected in 2007; spinetail devilray, manta ray, whale shark, deepwater nurse shark and basking shark in 2010; and oceanic whitetip shark in 2013.

Under-reporting of protected species by commercial fishers introduces a major bias into estimates of fishery interactions (Francis & Lyon 2012), but good observer coverage can go a long way to overcoming these biases. Observer coverage has been reasonably good over the last decade (2002–03 to 2012–13) or more in some large valuable fisheries (e.g., trawl fisheries for hoki (9.3–38.7% per year) and orange roughy (11.6–43.9% per year), and on licensed foreign fishing vessels (e.g., in the southern bluefin longline fishery; 32.3–56.6% per year). Some trawl fisheries around southern New Zealand and skipjack tuna purse seine fisheries in northern New Zealand also had reasonable coverage over about the last decade (southern blue whiting (25.2–99.9% per year), squid (12.9–85.9% per year) and tuna (12–29.1% per year 2005–06 to 2012–13), providing good information on captures of basking sharks, white sharks and spinetail devilrays. However, observer coverage has not always been representative of the spatial and temporal distribution of these fisheries. Inshore fisheries, notably set-net, bottom-longline and trawl fisheries, have received only sparse observer coverage. These fisheries may have unobserved and unrecorded mortality of some protected species, especially basking shark, white shark and deepwater nurse shark.

BASKING SHARK

Basking sharks are frequently taken as bycatch around southern New Zealand (Francis & Lyon 2012). The main capture locations are 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 also occurred around North Island but were relatively uncommon (Francis & Duffy 2002, Francis & Sutton 2012).

Most basking shark records came from trawl fisheries. The sharks were caught mainly by vessels targeting barracouta and hoki off east coast South Island, hoki off west coast South Island, and arrow squid off Southland-Auckland Island. Basking sharks are also caught in set nets (Francis & Duffy 2002) but have rarely been reported by fishers since they were protected in 2010 (Francis & Lyon 2012). The observer coverage of this fleet has been low, so the set net bycatch cannot be quantified. Basking sharks are rarely entangled in surface longlines (Francis & Duffy 2002).

Table 10.1: Recreational harvest estimates for New Zealand chondrichthyan species for the 2011–12 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. (2014).

Species	Mgmt class	Fishers (n)	Events (n)	Harvest (n)	CV	Mean weight (kg)	Harvest (t)	CV
Rig	QMS	159	241	47 718	0.14	1.09	52.05	0.14
School Shark	QMS	95	160	30 555	0.17	-	-	-
Spiny Dogfish Shark	QMS	97	119	22 200	0.19	1.02	22.60	0.19
Stingray		46	59	11 053	0.40	-	-	-
Elephant Fish	QMS	24	47	6 198	0.34	-	-	-
Sand Shark		10	18	3 719	0.54	-	-	-
Hammerhead Shark		10	12	1 429	0.34	-	-	-
Bronze Whaler Shark		5	5	570	0.52	-	-	-
Mako Shark	QMS	5	6	529	0.51	-	-	-
Carpet Shark		3	5	452	0.67	-	-	-

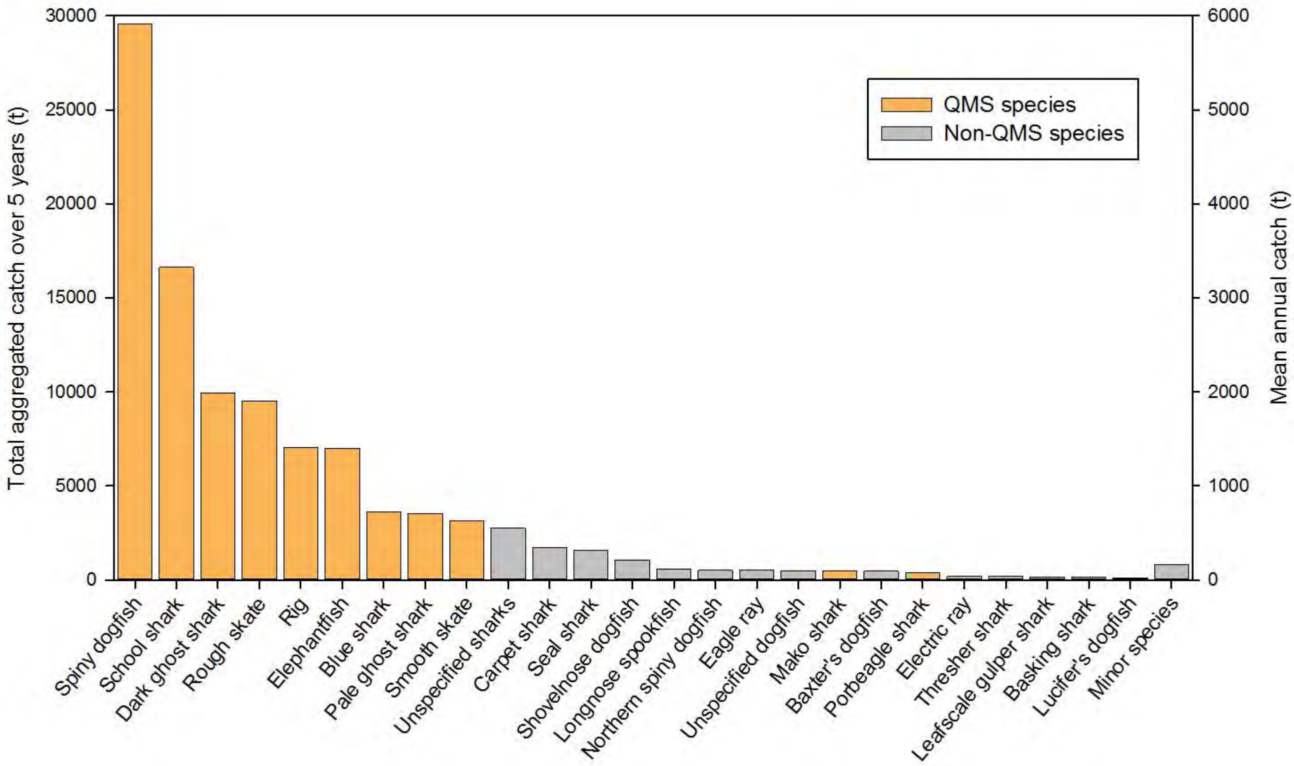

Figure 10.1: Reported total catches (landings, discards and live releases) for chondrichthyan species aggregated across 2009–10 to 2013–14. The average annual catches are shown on the right axis. Source: Ministry for Primary Industries catch-effort database.

Table 10.2: Reported total catches (tonnes, including discards and live releases) for chondrichthyan species from 2009–10 to 2013–14, arranged in descending order of total catch. Only species with more than 5 t of aggregated catch are included. The management class is also shown (blanks indicate Non-QMS species). Source: Ministry for Primary Industries catch-effort database. Note: Catches of QMS species differ from landings in Table 10.4 because they include discards and releases, and came from a different source. * Total five-year catches of protected species (basking shark and spinetail devilray) were estimated and scaled up from numbers of animals reported on protected species forms for the four years 2010–11 to 2013–14 (see text for details).

Species	Mgmt class	Code	2009-10	2010-11	2011-12	2012-13	2013-14	Total
Spiny dogfish	QMS	SPD	6626	6250	5704	5020	5945	29546
School shark	QMS	SCH	3389	3618	3315	3165	3144	16631
Dark ghost shark	QMS	GSH	2070	2326	2095	1712	1760	9963
Rough skate	QMS	RSK	1961	1937	1553	1919	2147	9517
Rig	QMS	SPO	1439	1457	1445	1311	1394	7045
Elephantfish	QMS	ELE	1386	1412	1382	1427	1401	7008
Blue shark	QMS	BWS	746	804	1054	740	302	3646
Pale ghost shark	QMS	GSP	799	632	695	700	719	3545
Smooth skate	QMS	SSK	581	649	580	646	708	3164
Unspecified sharks		OSD	609	597	697	586	275	2763
Carpet shark		CAR	296	349	336	340	396	1717
Seal shark		BSH	386	325	277	315	283	1587
Shovelnose dogfish		SND	192	186	145	182	367	1072
Longnose spookfish		LCH	131	97	101	117	126	572
Northern spiny dogfish		NSD	88	123	102	93	110	516
Eagle ray		EGR	81	105	108	92	130	515
Unspecified dogfish		DWD	234	98	78	35	63	508
Mako shark	QMS	MAK	76	95	160	87	56	474
Baxter's dogfish		ETB	46	47	30	41	303	467
Porbeagle shark	QMS	POS	68	77	60	94	100	400
Electric ray		ERA	30	37	38	41	48	194
Thresher shark		THR	30	38	38	37	44	187
Leafscale gulper shark		CSQ	20	14	9	30	96	169
Basking shark	Protected	BSK						165*
Lucifer's dogfish		ETL	26	17	25	32	21	121
Broadnose sevengill shark		SEV	17	18	19	20	21	95
Slender smoothhound		SSH	5	27	10	35	10	87
Short-tailed stingray		BRA	11	16	13	14	29	83
Bronze whaler shark		BWH	18	14	16	11	13	72
Long-tailed stingray		WRA	10	9	12	13	20	64
Hammerhead shark	Non-target	HHS	8	15	13	10	11	56
Unspecified stingray		STR	8	20	9	3	8	48
Longnose velvet dogfish		CYP	0	0	0	8	38	46
Giant chimaera		CHG	1	6	19	13	3	42
Deepwater spiny skate		DSK	11	13	0	8	1	33
Prickly dogfish		PDG	6	7	4	4	4	25
Spinetail devilray	Protected	MJA						24*
Unspecified skates		OSK	2	1	2	10	7	22
Unspecified rays		RAY	4	1	3	12	1	21
Unspecified chimaeras		CHI	2	11	1	2	2	17
Owston's dogfish		CYO	1	3	3	3	3	13
Longnose deepsea skate		PSK	7	2	1	1	1	11
Numbfish		BER	1	3	2	3	1	10
Plunket's shark		PLS	0	0	0	3	7	10
Sixgill shark		HEX	0	0	0	4	3	7
Softnose skate		LSK	2	1	1	1	1	5

Table 10.3: Percentages of reported total catches that were landed, discarded and released alive for chondrichthyan species from 2012–13 to 2013–14, arranged in descending order of total catch. Note: Species order differs from that in Table 10.2 because the two tables cover different time periods. Protected species are not included. Source: Ministry for Primary Industries catch-effort database.

Species	Code	Catch (t)	Landed %	Discarded %	Released %
Spiny dogfish	SPD	10965.0	39.0	61.0	0.0
School shark	SCH	6309.5	99.7	0.1	0.2
Rough skate	RSK	4065.9	97.9	0.2	1.9
Dark ghost shark	GSH	3471.6	97.8	2.2	0.0
Elephantfish	ELE	2828.6	100.0	0.0	0.0
Rig	SPO	2705.3	99.1	0.0	0.9
Pale ghost shark	GSP	1419.1	99.5	0.5	0.0
Smooth skate	SSK	1354.0	89.3	0.2	10.5
Blue shark	BWS	1042.7	75.8	4.6	19.6
Unspecified sharks	OSD	860.6	53.5	46.5	0.0
Carpet shark	CAR	735.4	10.6	89.4	0.0
Seal shark	BSH	598.5	75.4	24.6	0.0
Shovelnose dogfish	SND	549.8	71.0	29.0	0.0
Baxter's dogfish	ETB	343.2	84.3	15.7	0.0
Longnose spookfish	LCH	242.5	85.8	14.2	0.0
Eagle ray	EGR	222.1	62.5	37.5	0.0
Northern spiny dogfish	NSD	203.4	62.0	38.0	0.0
Porbeagle shark	POS	194.2	70.6	6.1	23.3
Mako shark	MAK	143.0	89.5	0.8	9.7
Leafscale gulper shark	CSQ	125.7	54.3	45.7	0.0
Unspecified dogfish	DWD	97.9	73.1	26.9	0.0
Electric ray	ERA	89.3	9.2	90.8	0.0
Thresher shark	THR	80.3	43.2	56.8	0.0
Lucifer's dogfish	ETL	52.7	50.2	49.8	0.0
Longnose velvet dogfish	CYP	45.9	76.9	23.1	0.0
Slender smoothhound	SSH	44.5	23.5	76.5	0.0
Short-tailed stingray	BRA	43.3	2.7	97.3	0.0
Broadnose sevengill shark	SEV	41.0	57.2	42.8	0.0
Long-tailed stingray	WRA	32.7	10.3	89.7	0.0
Bronze whaler shark	BWH	23.1	84.6	15.4	0.0
Hammerhead shark	HHS	20.9	95.8	4.2	0.0
Unspecified skates	OSK	16.9	12.2	87.8	0.0
Giant chimaera	CHG	16.6	20.5	79.5	0.0
Unspecified rays	RAY	13.5	1.5	98.5	0.0
Unspecified stingray	STR	10.5	3.8	96.2	0.0
Plunket's shark	PLS	10.3	19.3	80.7	0.0
Deepwater spiny skate	DSK	9.0	3.6	96.4	0.0
Prickly dogfish	PDG	7.9	9.3	90.7	0.0
Sixgill shark	HEX	6.6	6.3	93.7	0.0
Owston's dogfish	CYO	5.6	22.0	78.0	0.0
Numbfish	BER	4.1	0.9	99.1	0.0
Unspecified chimaeras	CHI	4.0	0.4	99.6	0.0
Longnose deepsea skate	PSK	1.6	1.5	98.5	0.0
Unspecified catshark	APR	1.4	86.9	13.1	0.0
Softnose skate	LSK	1.4	11.6	88.4	0.0

Table 10.4: TACCs and 2013–14 landings (tonnes) of the eleven chondrichthyans managed under the QMS. Also shown are the date of entry of each species into the QMS, and date of addition to Schedule 6 of the Fisheries Act that allows release of fish into the sea. Source: Monthly Harvest Returns (Ministry for Primary Industries 2014, 2015). Note: Landings differ from the catches in previous table because the latter include discards and releases, and came from a different source.

Species	Code	TACC (tonnes)	2013-14 landings	Entry into QMS	Addition to Schedule 6
Spiny dogfish	SPD	12660	5864	2004	2004
School shark	SCH	3436	3136	1986	2013
Rough skate	RSK	1986	2122	2003	2003
Dark ghost shark	GSH	3047	1817	1998	
Elephantfish	ELE	1304	1394	1986	
Rig	SPO	1941	1386	1986	2012
Pale ghost shark	GSP	1780	727	1999	
Smooth skate	SSK	849	641	2003	2003
Blue shark	BWS	1860	117	2004	2004
Porbeagle shark	POS	110	71	2004	2004
Mako shark	MAK	200	44	2004	2004

Most additional commercial records came from the early 2000s, but reporting rates appeared to be very low before 2000 (Francis & Lyon 2012). 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, or a relatively high abundance of sharks in the late 1980s and early 1990s (Francis & Sutton 2012).

Annual catch weights up to 2009–10 reported by commercial fishers ranged from 3 t to 150 t per year (Figure 10.3). Catch weights before 1999–2000 were undoubtedly under-reported. Since 2010, protected species bycatch has been reported on Non-Fish / Protected Species Catch Returns (NF/PSCR). Over the four fishing years 2010–11 to 2013–14, 37 basking sharks were reported on those forms. At an average weight of 3–4 t per shark (estimated weights from observers), this represents about 110–150 t of total catch, or 27–37 t per year. Few sharks were returned to the sea alive, and even fewer were likely to have survived their release.

WHITE SHARK

White 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). Regions with multiple captures included the west coast South Island off Hokitika, the southern edge of the Stewart-Snares Shelf, and the Auckland Islands Shelf. White sharks were mainly caught in FMAs 1, 5, 6 and 7.

Most white shark records came from trawl and set-net fisheries with few captures reported from surface and bottom longlines. Observer coverage of the set-net and bottom-longline fleet has been low, so the bycatch in these fisheries is likely to have been underestimated.

Three white sharks observed on surface longlines were recorded as struck off the line or lost. One white shark observed caught in a set net in 2009 was retained, whereas another shark was released alive. The life status of sharks observed caught on bottom longlines and in trawls was never recorded.

A maximum of 6.3 t was reported landed in 1990, but catches reported in other years have been low (and often zero). Catches of white sharks are considered to be under-reported.

WHALE SHARK

No captures of whale sharks have been reported by fishers or observers in New Zealand waters (Francis & Lyon 2012). However, a single individual was caught by a coastal trawler off South Canterbury in the late 1970s (as communicated to C. Duffy in Duffy 2005). This is exceptional, as whale sharks are typically only seen in north-eastern North Island waters during summer and are rare (Duffy 2002).

DEEPWATER NURSE SHARK (SMALLTOOTH SANDTIGER SHARK)

Deepwater nurse sharks 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 owing to use 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 (Francis & Lyon 2012).

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 Sealife Aquarium from the mid-1980s to the early 2000s, but all of the sharks died and the practice was discontinued.

SPINETAIL DEVILRAY AND MANTA RAY

Spinetail devilrays and manta rays occur mainly in north-eastern North Island waters during summer (Duffy & Abbott 2003). Most if not all mobulid rays reported caught in commercial fisheries were likely to have been spinetail devilrays (Paulin et al. 1982); no manta rays have been confirmed caught in New Zealand waters (Duffy 2005, Jones & Francis 2012). However, it is possible that manta rays are occasionally caught in purse seines 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) have not confirmed any captures.

All commercial and observer records of mobulid rays were from the northern North Island in FMAs 1 and 9, and most records came from purse seine vessels (Francis & Lyon 2012, Jones & Francis 2012). 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 devilrays 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 devilray bycatch in purse seine catches was documented between 1975 and 1981 by Paulin et al. (1982). All observed devilrays 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 seines was not recorded. Over the four fishing years 2010–11 to 2013–14, 153 spinetail devilrays 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 longliners, 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. However, most of them are rarely caught, rarely reported, or both). The main species known to be caught by commercial fishers can be grouped into five categories: inshore rays, inshore sharks, deepwater chimaeras, deepwater sharks and deepwater skates (Table 10.5). No analysis has been done of the interactions of most of these species with fisheries, but the presumed important fishing methods that catch these species are indicated in Table 10.5.

Inshore rays and sharks are caught by a variety of fishing methods. Closures of strips of 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, and their habitats and nursery areas. 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 of 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 (Table 10.5; 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, as 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, or processed for their fins and livers (Blackwell 2010). Catches of seal shark and shovelnose dogfish increased through the early 1990s, peaked in the early 2000s, and then declined, but these increases may have been affected by improved identification and reporting of deepwater shark catches (Blackwell 2010; Table 10.5). Historical data are available from the MPI Observer Programme (Figure 10.2), but coverage of the distribution of deepwater sharks has been unrepresentative.

Some species that are not caught or reported in quantities sufficient to be included in Table 10.5 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.

Table 10.5: Main Non-QMS chondrichthyans caught by commercial fishers, classified by species group and depth range. Only species with more than 5 t of aggregated catch between 2009–10 and 2013–14 are included. The main fishing methods thought to catch these species are also indicated. Source: M. Francis, pers. comm.

			Method		
Species group	Species	Code	Trawl	Bottom longline	Set net
Inshore rays	Eagle ray	EGR	+	+	+
	Electric ray	ERA	+	+	+
	Long-tailed stingray	WRA	+	+	+
	Short-tailed stingray	BRA	+	+	+
	Unspecified stingray	STR	+	+	+
	Unspecified rays	RAY	+	+	+
Inshore sharks	Bronze whaler shark	BWH		+	+
	Carpet shark	CAR	+	+	+
	Hammerhead shark	HHS		+	+
	Broadnose sevengill shark	SEV		+	+
	Thresher shark	THR	+		+
	Basking shark	BSK	+		+
Deepwater chimaeras	Giant chimaera	CHG	+		
	Longnose spookfish	LCH	+		
	Unspecified chimaeras	CHI	+		
Deepwater sharks	Baxter's dogfish	ETB	+		
	Leafscale gulper shark	CSQ	+		
	Longnose velvet dogfish	CYP	+		
	Lucifer's dogfish	ETL	+		
	Northern spiny dogfish	NSD	+		
	Owston's dogfish	CYO	+		
	Plunket's shark	PLS	+		
	Prickly dogfish	PDG	+		
	Seal shark	BSH	+		
	Shovelnose dogfish	SND	+		
	Sixgill shark	HEX	+		
	Slender smoothhound	SSH	+		
	Unspecified dogfish	DWD	+		
	Deepwater spiny skate	DSK	+		
	Longnose deepsea skate	PSK	+		
Deepwater skates	Numbfish	BER	+		
	Softnose skate	LSK	+		
	Unspecified skates	OSK	+		

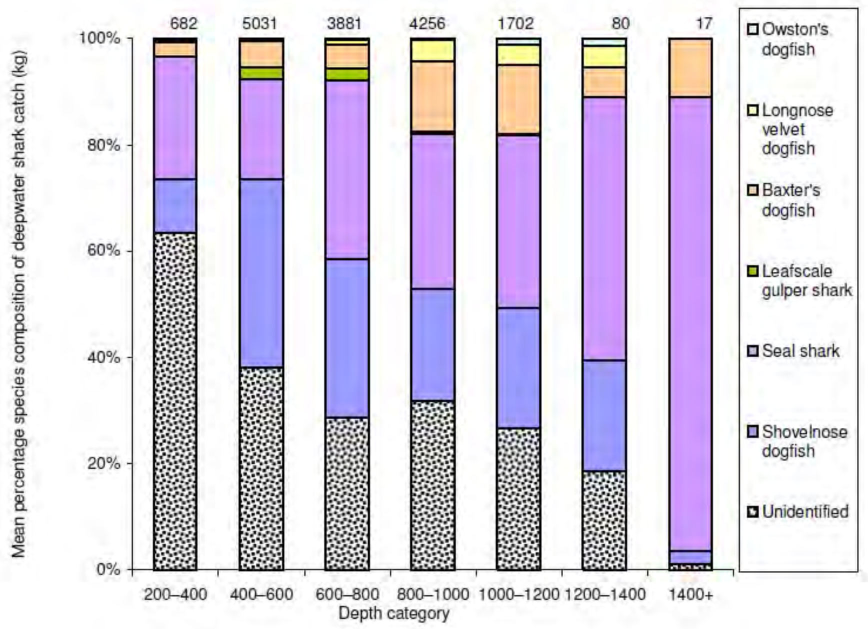

Figure 10.2: Mean catch composition of deepwater chondrichthyans reported from the Observer Programme database, all years 2001–02 to 2005–06, by major depth category (number of observations shown above bars). Source: Blackwell (2010).

10.5 INDICATORS AND TRENDS

RISK ASSESSMENT

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. This is expected to be followed by a more quantitative (Level 2) RA prior to the next review of the NPOA-Sharks in 2017.

The qualitative RA used a modified Scale Intensity Consequence Analysis approach (Ford et al. 2015). Before the workshop, 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. 2015). Results were reported within the three management classes of chondrichthyans – QMS, Non-QMS and Protected species.

Carpet shark (Non-QMS) and rough skate (QMS) attained the highest total risk scores, and the highest scoring protected species was basking shark (Appendix 10.3). The panel did not consider available information suggested that commercial fishing is currently causing, or in the near future could cause, serious unsustainable impacts to any sharks, rays or chimaera population examined. However, out of the 84 taxa considered, the panel had low confidence in the risk scores for 43 taxa and consensus was not reached for 3 taxa.

The risk assessment was designed to help prioritise actions to conserve chondrichthyans, noting that protected species are also given priority under the NPOA-Sharks (2013). The panel made a number of recommendations for research on high-risk or protected species. These included more use of existing data , grooming or analysis to improve inputs to assessment scores, better taxonomy and education to improve the identification of chondrichthyans, and collection of more biological information to improve our understanding of productivity (especially the ability of a taxon to recover from fishing impacts) (Ford et al. 2015).

QMS SPECIES

Standardised CPUE analyses have been carried out to monitor trends in the relative abundance of some stocks of 7 of the 11 QMS chondrichthyans species (rig, school sharks, elephantfish, pale ghost shark, blue shark, porbeagle shark and mako shark) (Tables 10.6, 10.7). For all 20 of the stocks that are monitored, stock size is stable or increasing in recent years.

For blue, porbeagle and mako sharks, other abundance indicators have been developed in addition to standardized 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 (hitch 2014). None of the indicators for the period 2005−13 suggested that any of the three shark species were declining. In fact, some of the indicators suggested positive trends for all three species.

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 (Tables 10.6, 10.7). For 20 out of 22 species/FMA combinations, abundance is stable or increasing in recent years; however smooth skate and pale ghost shark in FMA 4 have downward trends.

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 Island SA) are shown in Table 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 sharks were observed in 49 165 tows in 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 may be attributable to targeting by Japanese vessels (Francis & Sutton 2012). However, the very low (often zero) CPUE since then, 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).

Table 10.6: Trends in abundance of QMS species monitored by standardised CPUE analysis and trawl surveys. Changes in trends through time are indicated by forward slashes, and multiple substocks or multiple indices within QMAs are separated by commas. Blanks, none or unreliable. Source: Ministry for Primary Industries (2014, 2015) unless otherwise indicated. 'Recent years' refers to the last five years, but may be longer for long time series.

CPUE indices		QMA 1	QMA 2	QMA 3	QMA 4	QMA 5	QMA 6	QMA 7	QMA 8	Source
Rig	SPO	Nil, Down/ Nil, Nil	Up/Nil/Up	Nil				Down/Up, Nil	Down/Up, Nil	
School shark	SCH	Nil, Up	Nil	Nil	Nil	Nil		Nil/Up	Nil	
Elephantfish	ELE			Nil		Nil		Nil		
Pale ghost shark	GSP	Nil				Up				MacGibbon & Fu (2013)
Blue shark	BWS	Up								
Porbeagle shark	POS	Up								
Mako shark	MAK	Up								
Trawl survey indices				FMA 3	FMA 4	FMA 5	FMA 6	FMA 7		
Rig	SPO			Up/Nil				Nil		
School shark	SCH			Nil	Up					FMA 4: O'Driscoll et al. (2011)
Spiny dogfish	SPD			Nil	Nil	Nil	Nil	Nil		FMAs 5&6: Bagley et al. (2013)
Elephantfish	ELE			Nil						
Rough skate	RSK			Up/Nil	Nil			Down/Up		
Smooth skate	SSK			Up/Nil	Up/Down			Down/Up		
Dark ghost shark	GSH			Up	Up/Nil	Nil		Nil		
Pale ghost shark	GSP				Nil/Down	Up				
Legend:										
(green)	Trend up in recent years									
(grey)	Stable in recent years									
(red)	Trend down in recent years									

NON-QMS SPECIES

Some Non-QMS deepwater chondrichthyans have been monitored by trawl surveys on the Chatham Rise and subantarctic (Campbell Plateau) over a period of almost two decades. Trends in relative abundance indices and mean length were provided by O'Driscoll et al. (2011) and Bagley et al. (2013) and are summarised in Table 10.7. These survey series covered only a small part of the known distributions of these species, and it is not known how representative the results are. Most species showed no trends in biomass or mean length. However, on the Chatham Rise prickly dogfish increased and then declined. In the Subantarctic, leafscale gulper shark, Baxter's dogfish and shovelnose dogfish all increased.

Anderson (2013) analysed trends in bycatch quantities caught in eight deepwater trawl fisheries from 1990–91 to 2010–11. Some species showed consistent declines or increases across six or more of the eight fisheries. Deepsea skates, Baxter's dogfish, and lucifer dogfish all increased while two groupings (unspecified sharks and unspecified skates) decreased. These trends appear to be a direct result of better reporting of deepwater sharks and skates by species code rather than by an unspecified generic code, and should not be interpreted as trends in abundance of the species.

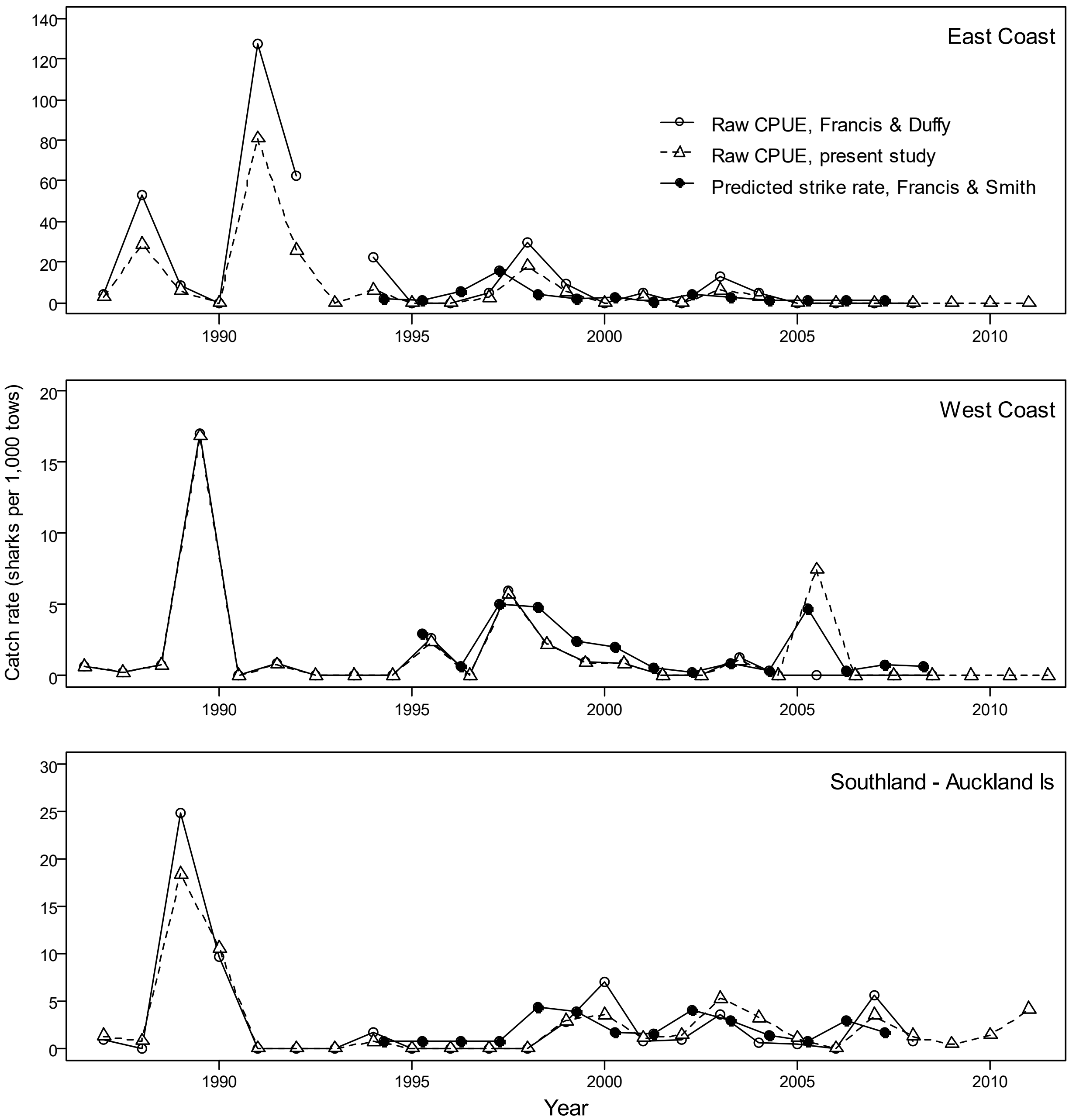

Figure 10.3: Basking shark catch rate indices for three fishery areas. For raw CPUE indices, years are calendar years for West Coast and July–June years (labelled as the greater of the two years) for East Coast and Southland-Auckland Is. For predicted strike rate, years are fishing years (labelled as the greater of the two years). Source: Francis & Sutton (2012).

Table 10.7: Trends in relative biomass and mean length determined from time series of research bottom trawl surveys of the Chatham Rise and the subantarctic (Campbell Plateau). Sources: O'Driscoll et al. (2011), Bagley et al. (2013).

Code	Species	Quality of biomass estimate	Biomass trend	Mean length trend
Chatham Rise, 1992-2010				
BSH	Seal shark	moderate	no change	
CYP	Longnose velvet dogfish	poor		no change
ETB	Baxter's dogfish	moderate	no change	no change
ETL	Lucifer's dogfish	very good	no change	no change
GSH	Dark ghost shark	very good	increase	decrease
GSP	Pale ghost shark	very good	no change	decrease
LCH	Longnose spookfish	very good	no change	no change
PDG	Prickly dogfish	moderate	increase then decrease	
SCH	School shark	moderate	increase	
SKA	Unspecified skates	good	no change	
SND	Shovelnose dogfish	good	no change	no change
SPD	Spiny dogfish	very good	increase	increase then decrease
SSK	Smooth skate	good	increase	no change
Subantarctic, 1991-1993 and 2000-2009				
BTH	Bluntnose deepwater skates	moderate	no change	
CSQ	Leafscale gulper shark	moderate	increase	increase
CYP	Longnose velvet dogfish	moderate	no change	no change
ETB	Baxter's dogfish	good	increase	no change
ETL	Lucifer's dogfish	good	no change	decrease
GSH	Dark ghost shark	poor		no change
GSP	Pale ghost shark	very good	no change	no change
LCH	Longnose spookfish	good	no change	no change
SND	Shovelnose dogfish	good	increase	no change
SPD	Spiny dogfish	good	no change	decrease

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10.7 APPENDICES

Appendix 10.1: List of New Zealand chondrichthyans, with details of their fisheries management classification, and IUCN and Department of Conservation threat classes. IUCN threat classes: EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern; DD, Data Deficient. The regional red list class is given for *Squalus acanthias* (LC) because it differs from the global class of VU. DOC threat classes: GD, Gradual Decline; RR, Range restricted; SP, Sparse; NOT, Not Threatened; MI, Migrant; VA, Vagrant. DOC qualifiers: CD, Conservation Dependent; DP, Data Poor; RC, Recovering; SO, Secure Overseas; TO, Threatened Overseas. Sources: IUCN Redlist classes as at July 2013 (L. Harrison, Shark Specialist Group IUCN, pers. comm.); DOC threat classes 2005 (Hitchmough et al. 2007). [Continued on next pages]

Group	Family	Species	Common name	Code	Manage-ment class	IUCN redlist class	DoC threat class	DoC qualifier
Chimaera	Callorhinchidae	*Callorhinchus milii* Bory de St Vincent, 1823	Elephantfish	ELE	QMS	LC	NOT	CD,RC
Chimaera	Rhinochimaeridae	*Harriotta haeckeli* Karrer, 1972	Smallspine spookfish	HHA	Non-QMS	DD	NOT	DP,SO
Chimaera	Rhinochimaeridae	*Harriotta raleighana* Goode & Bean, 1895	Longnose spookfish	LCH	Non-QMS	LC	NOT	SO
Chimaera	Rhinochimaeridae	*Rhinochimaera pacifica* (Mitsukuri, 1895)	Pacific spookfish	RCH	Non-QMS	LC	NOT	SO
Chimaera	Chimaeridae	*Chimaera carophila* Kemper, Ebert, Naylor & Didier 2014	Brown chimaera, longspine chimaera	CHP	Non-QMS		NOT	
Chimaera	Chimaeridae	*Chimaera lignaria* Didier, 2002	Purple chimaera, giant chimaera	CHG	Non-QMS	DD	NOT	SO
Chimaera	Chimaeridae	*Chimaera panthera* Didier, 1998	Leopard chimaera	CPN	Non-QMS	DD	NOT	
Chimaera	Chimaeridae	*Hydrolagus bemisi* Didier, 2002	Pale ghost shark	GSP	QMS	LC	NOT	
Chimaera	Chimaeridae	*Hydrolagus homonycteris* Didier 2008	Black ghost shark	HYB	Non-QMS	DD	NOT	SO
Chimaera	Chimaeridae	*Hydrolagus novaezealandiae* (Fowler, 1910)	Dark ghost shark	GSH	QMS	LC	NOT	
Chimaera	Chimaeridae	*Hydrolagus trolli* Didier and Seret, 2002	Pointynose blue ghost shark	HYP	Non-QMS	DD	NOT	SO
Chimaera	Chimaeridae	*Hydrolagus* sp. D [Didier]	Giant black ghost shark	HGB	Non-QMS		DD	
Shark	Chlamydoselachidae	*Chlamydoselachus anguineus* Garman, 1884	Frill shark	FRS	Non-QMS	NT	SP	DP,SO
Shark	Hexanchidae	*Heptranchias perlo* (Bonnaterre, 1788)	Sharpnose sevengill shark	HEP	Non-target	NT	SP	DP,SO
Shark	Hexanchidae	*Hexanchus griseus* (Bonnaterre, 1788)	Sixgill shark	HEX	Non-QMS	NT	SP	DP,SO
Shark	Hexanchidae	*Notorynchus cepedianus* (Peron, 1807)	Broadnose sevengill shark	SEV	Non-QMS	DD	NOT	DP,SO
Shark	Echinorhinidae	*Echinorhinus brucus* (Bonnaterre, 1788)	Bramble shark	BRS	Non-QMS	DD	SP	DP,SO
Shark	Echinorhinidae	*Echinorhinus cookei* Pietschmann, 1928	Prickly shark	ECO	Non-QMS	NT	SP	DP,SO
Shark	Squalidae	*Cirrhigaleus australis* White, Last & Stevens, 2007	Southern mandarin dogfish	MSH	Non-QMS	DD	SP	DP,TO
Shark	Squalidae	*Squalus acanthias* Linnaeus, 1758	Spiny dogfish	SPD	QMS	LC	NOT	SO
Shark	Squalidae	*Squalus griffini* Phillipps, 1931	Northern spiny dogfish	NSD	Non-QMS	LC	NOT	SO
Shark	Squalidae	*Squalus raoulensis* Duffy & Last, 2007	Kermadec spiny dogfish		Non-QMS	LC	DD	
Shark	Squalidae	*Squalus* sp. 5	Green-eye dogfish		Non-QMS		DD	
Shark	Centrophoridae	*Centrophorus harrissoni* McCulloch, 1915	Harrisson's dogfish		Non-QMS	EN	DD	TO
Shark	Centrophoridae	*Centrophorus squamosus* (Bonnaterre, 1788)	Leafscale gulper shark	CSQ	Non-QMS	VU	NOT	
Shark	Centrophoridae	*Deania calcea* (Lowe, 1839)	Shovelnose dogfish	SND	Non-QMS	LC	NOT	SO
Shark	Centrophoridae	*Deania histricosa* (Garman, 1906)	Rough longnose dogfish	SNR	Non-QMS	DD		
Shark	Centrophoridae	*Deania quadrispinosa* (McCulloch, 1915)	Longsnout dogfish	DEQ	Non-QMS	NT	DD	SO
Shark	Etmopteridae	*Centroscyllium* sp. cf. *kamoharai*	Fragile dogfish		Non-QMS		DD	
Shark	Etmopteridae	*Etmopterus granulosus* (Günther, 1880)	Baxter's dogfish	ETB	Non-QMS	LC	NOT	SO
Shark	Etmopteridae	*Etmopterus lucifer* Jordan & Snyder, 1902	Lucifer's dogfish	ETL	Non-QMS	LC	NOT	SO
Shark	Etmopteridae	*Etmopterus molleri* (Whitley, 1939)	Moller's lantern shark	EMO	Non-QMS	LC	NOT	SO
Shark	Etmopteridae	*Etmopterus pusillus* (Lowe, 1839)	Smooth lantern shark	ETP	Non-QMS	LC	SP	DP,SO
Shark	Etmopteridae	*Etmopterus* cf. *unicolor*	Bristled lantern shark		Non-QMS		NOT	SO
Shark	Etmopteridae	*Etmopterus viator* Straube 2012	Blue-eye lantern shark	EVI	Non-QMS			

Appendix 10.1 [Continued]:

Group	Family	Species	Common name	Code	Manage-ment class	IUCN redlist class	DoC threat class	DoC qualifier
Shark	Somniosidae	*Centroscymnus coelolepis* Bocage & Capello, 1864	Portuguese dogfish	CYL	Non-QMS	NT	NOT	
Shark	Somniosidae	*Centroscymnus owstonii* Garman, 1906	Owston's dogfish	CYO	Non-QMS	LC	NOT	
Shark	Somniosidae	*Centroselachus crepidater* (Bocage & Capello, 1864)	Longnose velvet dogfish	CYP	Non-QMS	LC	NOT	
Shark	Somniosidae	*Scymnodalatias albicauda* Taniuchi & Garrick, 1986	Whitetail dogfish	SLB	Non-QMS	DD	SP	DP,SO
Shark	Somniosidae	*Scymnodalatias sherwoodi* (Archey, 1921)	Sherwood's dogfish	SHE	Non-QMS	DD	SP	
Shark	Somniosidae	*Scymnodon macracanthus* Regan 1906	Roughskin dogfish		Non-QMS			
Shark	Somniosidae	*Scymnodon plunketi* (Waite, 1910)	Plunket's shark	PLS	Non-QMS	NT	NOT	
Shark	Somniosidae	*Scymnodon ringens* Bocage & Capello, 1864	Knifetooth dogfish	SRI	Non-QMS		DD	SO
Shark	Somniosidae	*Somniosus antarcticus* Whitley, 1939	Southern sleeper shark	SOP	Non-QMS	DD	SP	DP,SO
Shark	Somniosidae	*Somniosus longus* (Tanaka, 1912)	Little sleeper shark	SOM	Non-QMS	DD	DD	SO
Shark	Somniosidae	*Zameus squamulosus* (Günther, 1877)	Velvet dogfish	ZAS	Non-QMS	DD	SP	DP,SO
Shark	Oxynotidae	*Oxynotus bruniensis* (Ogilby, 1893)	Prickly dogfish	PDG	Non-QMS	DD	NOT	DP,SO
Shark	Dalatiidae	*Dalatias licha* (Bonnaterre, 1788)	Seal shark, black shark	BSH	Non-QMS	NT	NOT	SO
Shark	Dalatiidae	*Euprotomicrus bispinatus* (Quoy & Gaimard, 1824)	Pygmy shark	EBI	Non-QMS	LC	NOT	SO
Shark	Dalatiidae	*Isistius brasiliensis* (Quoy & Gaimard, 1824)	Cookie cutter shark	IBR	Non-QMS	LC	NOT	SO
Shark	Heterodontidae	*Heterodontus portusjacksoni* (Meyer, 1793)	Port Jackson shark	PJS	Non-QMS	LC	VA	SO
Shark	Rhincodontidae	*Rhincodon typus* (Smith, 1828)	Whale shark	WSH	Protected	VU	MI	SO
Shark	Odontaspidae	*Odontaspis ferox* (Risso, 1810)	Deepwater (smalltooth) sand tiger shark	ODO	Protected	VU	SP	TO
Shark	Pseudocarchariidae	*Pseudocarcharias kamoharai* (Matsubara, 1936)	Crocodile shark.	CRC	Non-QMS	NT	DD	SO
Shark	Mitsukurinidae	*Mitsukurina owstoni* Jordan, 1898	Goblin shark	GOB	Non-QMS	LC	SP	DP,SO
Shark	Alopiidae	*Alopias superciliosus* (Lowe, 1839)	Bigeye thresher	BET	Non-QMS	VU	NOT	TO
Shark	Alopiidae	*Alopias vulpinus* (Bonnaterre, 1788)	Thresher shark	THR	Non-QMS	VU	NOT	TO
Shark	Cetorhinidae	*Cetorhinus maximus* (Gunnerus, 1765)	Basking shark	BSK	Protected	VU	GD	TO
Shark	Lamnidae	*Carcharodon carcharias* (Linnaeus, 1758)	White shark, white pointer	WPS	Protected	VU	GD	TO
Shark	Lamnidae	*Isurus oxyrinchus* Rafinesque, 1810	Mako shark, shortfin mako	MAK	QMS	VU	NOT	SO
Shark	Lamnidae	*Lamna nasus* (Bonnaterre, 1788)	Porbeagle shark	POS	QMS	VU	NOT	TO
Shark	Scyliorhinidae	*Apristurus ampliceps* Sasahara, Sato & Nakaya 2008	Roughskin cat shark	APR	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Apristurus* cf. *australis* Sato, Nakaya & Yorozu 2008	Pinocchio cat shark	APR	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Apristurus exsanguis* Sato, Nakaya and Stewart 1999	Pale catshark	APR	Non-QMS	LC	NOT	
Shark	Scyliorhinidae	*Apristurus melanoasper* Iglésias, Nakaya & Stehmann 2004	Fleshynose cat shark	APR	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Apristurus pinguis* Deng, Xiong & Zhan 1983	Cat shark	APR	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Apristurus sinensis* Chu & Hu 1981	Freckled cat shark	APR	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Apristurus* sp.	Cat shark	APR	Non-QMS		NOT	
Shark	Scyliorhinidae	*Bythaelurus dawsoni* (Springer, 1971)	Dawson's cat shark	DCS	Non-QMS	DD	NOT	
Shark	Scyliorhinidae	*Cephaloscyllium isabellum* (Bonnaterre, 1788)	Carpet shark	CAR	Non-QMS	LC	NOT	
Shark	Scyliorhinidae	*Cephaloscyllium* sp.	Swellshark		Non-QMS		DD	
Shark	Scyliorhinidae	*Parmaturus bigus* Seret & Last, 2007	Shorttail cat shark		Non-QMS	DD		
Shark	Scyliorhinidae	*Parmaturus macmillani* Hardy, 1985	McMillan's cat shark	PCS	Non-QMS	DD	DD	SO
Shark	Scyliorhinidae	*Parmaturus* sp.	Rough-backed cat shark		Non-QMS		DD	
Shark	Scyliorhinidae	*Parmaturus* sp.			Non-QMS			

Appendix 10.1 [Continued]:

Group	Family	Species	Common name	Code	Manage-ment class	IUCN redlist class	DoC threat class	DoC qualifier
Shark	Pseudotriakidae	*Gollum attenuatus* (Garrick, 1954)	Slender smooth hound	SSH	Non-QMS	LC	NOT	SO
Shark	Pseudotriakidae	*Pseudotriakis microdon* Capello, 1868	False cat shark	PMI	Non-QMS	DD	DD	SO
Shark	Triakidae	*Galeorhinus galeus* (Linnaeus, 1758)	School shark	SCH	QMS	VU	NOT	CD,TO
Shark	Triakidae	*Mustelus lenticulatus* Phillipps, 1932	Rig	SPO	QMS	LC	NOT	CD
Shark	Triakidae	*Mustelus* sp.	Kermadec Rig		Non-QMS		RR	SO
Shark	Carcharhinidae	*Carcharhinus brachyurus* (Günther, 1870)	Bronze whaler	BWH	Non-QMS	NT	NOT	SO
Shark	Carcharhinidae	*Carcharhinus galapagensis* (Snodgrass & Heller, 1905)	Galapagos shark	CGA	Non-QMS	NT	RR	SO
Shark	Carcharhinidae	*Carcharhinus longimanus* (Poey, 1861)	Oceanic whitetip shark	OWS	Protected	VU	MI	SO
Shark	Carcharhinidae	*Carcharhinus obscurus* (Le Sueur, 1818)	Dusky shark	DSH	Non-QMS	VU	MI	SO
Shark	Carcharhinidae	*Galeocerdo cuvier* (Peron & Le Sueur, 1822)	Tiger shark	TIS	Non-QMS	NT	MI	SO
Shark	Carcharhinidae	*Prionace glauca* (Linnaeus, 1758)	Blue shark	BWS	QMS	NT	NOT	SO
Shark	Sphyrnidae	*Sphyrna zygaena* (Linnaeus, 1758)	Hammerhead shark, smooth hammerhead	HHS	Non-target	VU	NOT	SO
Batoid	Narkidae	*Typhlonarke aysoni* (Hamilton, 1902)	Blind electric ray	TAY	Non-QMS	DD	NOT	DP
Batoid	Narkidae	*Typhlonarke tarakea* Phillipps, 1929	Oval electric ray	TTA	Non-QMS	DD	NOT	DP
Batoid	Torpedinidae	*Torpedo fairchildi* Hutton, 1872	Electric ray	ERA	Non-QMS	DD	NOT	
Batoid	Arhynchobatidae	*Arhynchobatis asperrimus* Waite, 1909	Longtail skate	LSK	Non-QMS	DD	NOT	
Batoid	Arhynchobatidae	*Bathyraja* cf. *eatonii*	Antarctic allometric skate	BEA	Non-QMS			
Batoid	Arhynchobatidae	*Bathyraja maccaini* Springer 1971	MacCain's skate	MCS	Non-QMS	NT		
Batoid	Arhynchobatidae	*Bathyraja richardsoni* (Garrick, 1961)	Richardson's skate	RIS	Non-QMS	LC	DD	SO
Batoid	Arhynchobatidae	*Bathyraja shuntovi* Dolganov, 1985	Longnose deepsea skate	PSK	Non-QMS	DD	NOT	
Batoid	Arhynchobatidae	*Bathyraja* sp.	Antarctic dwarf skate	BHY	Non-QMS			
Batoid	Arhynchobatidae	*Bathyraja* sp.	Blonde skate		Non-QMS			
Batoid	Arhynchobatidae	*Brochiraja albilabiata* Last & McEachran, 2006			Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja asperula* (Garrick & Paul, 1974)	Smooth deepsea skate	BTA	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja leviveneta* Last & McEachran, 2006			Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja microspinifera* Last & McEachran, 2006			Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Brochiraja spinifera* (Garrick & Paul, 1974)	Prickly deepsea skate	BTS	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Notoraja sapphira* Seret & Last 2009	Sapphire skate	BTH	Non-QMS	DD	DD	
Batoid	Arhynchobatidae	*Notoraja* [subgenus C] sp. A [Last & McEachran]		BTH	Non-QMS		DD	
Batoid	Arhynchobatidae	*Notoraja* [subgenus C] sp. B [Last & McEachran]		BTH	Non-QMS		DD	
Batoid	Arhynchobatidae	*Notoraja* [subgenus C] sp. C [Last & McEachran]		BTH	Non-QMS		DD	
Batoid	Arhynchobatidae	*Notoraja* [subgenus D] sp. A [Last & McEachran]		BTH	Non-QMS		DD	
Batoid	Rajidae	*Amblyraja georgiana* (Norman 1938)	Antarctic starry skate	SRR	Non-QMS	DD		
Batoid	Rajidae	*Amblyraja* cf. *hyperborea* (Collette, 1879)	Arctic skate	DSK	Non-QMS		NOT	
Batoid	Rajidae	*Dipturus innominatus* (Garrick & Paul, 1974)	Smooth skate	SSK	QMS	NT	NOT	CD
Batoid	Rajidae	*Zearaja nasuta* (Banks in Müller & Henle, 1841)	Rough skate	RSK	QMS	LC	NOT	
Batoid	Dasyatidae	*Dasyatis brevicaudata* (Hutton, 1875)	Shorttail stingray	BRA	Non-QMS	LC	NOT	SO
Batoid	Dasyatidae	*Dasyatis thetidis* Ogilby in Waite, 1899	Longtail stingray	WRA	Non-QMS	DD	NOT	SO
Batoid	Dasyatidae	*Pteroplatytrygon violacea* (Bonaparte, 1832)	Pelagic stingray	DAS	Non-QMS	LC	NOT	SO
Batoid	Myliobatidae	*Myliobatis tenuicaudatus* Hector, 1877	Eagle ray	EGR	Non-QMS	LC	NOT	SO
Batoid	Mobulidae	*Manta birostris* (Donndorff, 1798)	Manta ray	RMB	Protected	VU	MI	SO
Batoid	Mobulidae	*Mobula japanica* (Müller & Henle, 1841)	Spinetail devilray	MJA	Protected	NT	NOT	SO

Appendix 10.2: 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	●	●●	●●●	■	-
Elephant fish	ELE5	2014		●●	●●	●●	-
Elephant fish	ELE7	2015		●●	●●●	●●	-
Ghost shark - dark	GSH1, GSH2, GSH7, GSH8, GSH9	-					-
Ghost shark - dark	GSH3	-				●●	-
Ghost shark - dark	GSH4, GSH5, GSH6	-					-
Ghost shark - pale	GSP1, GSP5	2011		●●	●●●		-
Ghost shark - pale	GSP7	-					-
Mako shark*	MAK1	2014					TAC reduced in 2012
Porbeagle shark*	POS1	2014					TAC reduced in 2012
Rig	SPO1	2016					-
Rig	SPO2	2016	●	●●	●●●	●●	-
Rig	SPO3	2016	●●	●●●	●●●	■	-
Rig	SPO7, SPO8	2016	●●	●●●	●●●	●●●	-
School shark	SCH1, SCH2, SCH3, SCH4, SCH5, SCH7,	2014			●●●		-
Skate - rough	RSK1, RSK3, RSK7, RSK8	2007					-
Skate - smooth	SSK1, SSK3, SSK7, SSK8	2007					-
Spiny dogfish	SPD1, SPD8	-					-
Spiny dogfish	SPD3, SDP7	2009			●●		-
Spiny dogfish	SPD4	2009			●●		-
Spiny dogfish	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 2014, for the sixth consecutive year, three additional indicators of stock and fishery status have been analysed. The three new indicators are defined in the Harvest Strategy Standard for New Zealand Fisheries approved by the Minister of Fisheries on 24 October 2008.

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 (2015) 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 2014 data published by the Ministry for Primary Industries on its website (http://fs.fish.govt.nz/Page.aspx?pk=16&tk=478).

Appendix 10.3: QMS chondrichthyan risk assessment results from Ford et al. (2015). [Continued on next pages]

QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
6	3.5	21 - Rough skate	✓✓	✓✓
5	4	20 - Smooth skate	✓✓	✓✓
6	3	18 - Dark ghost shark	✓✓	✓✓
6	3	18 - Elephantfish	✓✓✓	✓✓
6	3	18 - Rig	✓✓✓	✓✓
6	3	18 - School shark	✓✓✓	✓✓
6	3	18 - Spiny dogfish	✓✓✓	✓✓
5	3	15 - Mako shark	✓✓✓	✓
5	3	15 - Pale Ghost Shark	✓✓	✗
5	3	15 - Porbeagle shark	✓✓✓	✓
4	3	12 - Blue shark	✓✓✓	✓✓

Figure 6: QMS Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 2 and Table 3). 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 in alphabetical order.

Appendix 10.3 [Continued]:

NON-QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
6	3.5	21 - Carpet Shark	✓✓	✓✓
5	4	20 - Baxters dogfish	✓✓	✓✓
5	4	20 - Seal shark	✓✓	✓
4.5	4	18 - Leafscale gulper shark	✓✓	✓✓
4	4.5	18 - Longnose velvet dogfish	✓✓	✓
4	4.5	18 - Plunkets shark	✓✓	✓
5	3.5	17.5 - Electric ray	✓	✓
5	3.5	17.5 - Shovelnose dogfish	✓✓	✓✓
4	4	16 - Blind electric ray	✓✓	✓
4	4	16 - Oval electric ray	✓✓	✓
4	4	16 - Owstons dogfish	✓✓	✓
3.5	4.5	15.75 - Dawsons catshark	✓✓	✓
4.5	3.5	15.75 Longnose spookfish	✓✓	✗
3.5	4	14 - Bronze whaler	✓✓	✗
4	3.5	14 - Longtail stingray	✓	✓✓
4	3.5	14 - Northern spiny dogfish	✓✓	✓
3.5	4	14 - Prickly dogfish	✓✓	✓
4	3.5	14 - Shorttail stingray	✓	✓✓
3.5	3.5	12.25 - Prickly deepsea skate	✓✓	✓
3.5	3.5	12.25 - Smooth deepsea skate	✓✓	✓
4	3	12 - Broadnose sevengill shark	✓✓	✓
3	4	12 - *Brochiraja* complex	✓	✓
3	4	12 - Brown chimaera	✓	✓✓
3	4	12 - Catsharks	✓	✓
3	4	12 - Deepwater spiny skate	✓	✓
4	3	12 - Hammerhead shark	✓✓	✓
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	✓✓	✓

Figure 7: Non-QMS Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 2 and Table 3). 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 taxa scored identical risk scores they are presented so that higher consequences are reported first and then in alphabetical order.

Appendix 10.3 [Continued]:

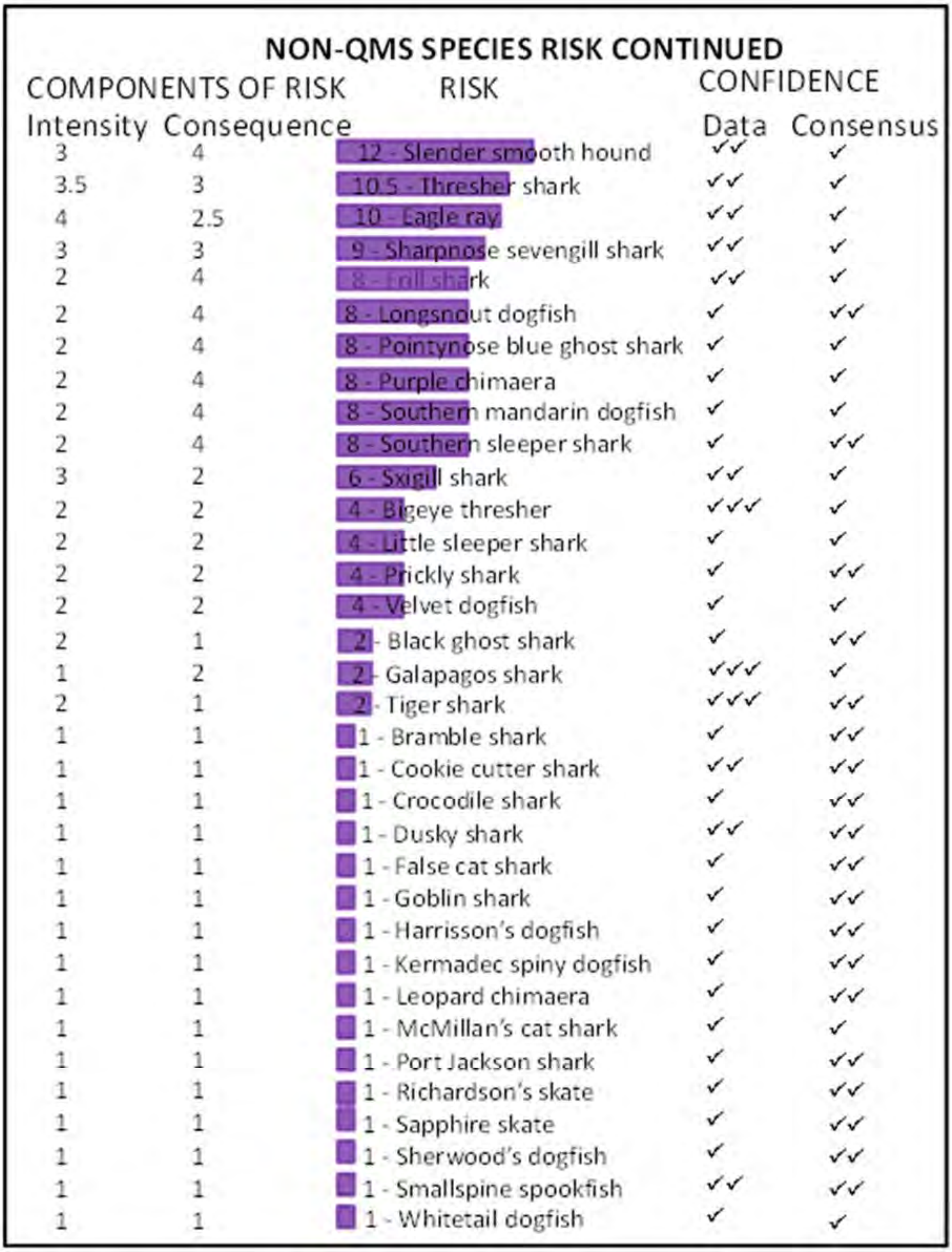


Figure 7 (continued): Non-QMS taxa risk scores (see the previous page for full legend details).

Appendix 10.3 [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	✓✓	✓
2	4	8 - Smalltoothed sandtiger	✓	✓
1	1	1 - Whale shark	✓✓✓	✓✓
1	1	1 - Oceanic whitetip shark	✓✓✓	✓✓
1	1	1 - Manta ray	✓✓	✓✓

Figure 8: Protected Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 2 and Table 3). 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 in alphabetical order.

THEME 3: BENTHIC IMPACTS

11 BENTHIC (SEABED) IMPACTS

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 seines and more or less static methods like bottom longline and potting are excluded in this version, as are fisheries outside the EEZ.
Area	All of the New Zealand Territorial Sea (TS) and Exclusive Economic Zone (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, trawling is most intense along the east coast of North Island, south of East Cape, and in Tasman and Golden Bays. Shellfish dredges probably have the greatest effect but their footprint is much smaller than that of bottom trawl fisheries and generally in shallow waters.
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). The need for (and opportunities presented by) better spatial information on inshore fisheries from finer scale reporting of fishing locations (including logbooks). Cumulative effects and interactions with other stressors (including existing effects, especially in the coastal zone, and climate change.
MPI research (current)	BEN2017-01 *Monitoring the trawl footprint for deepwater fisheries*; DAE2017-02 *Taxonomic identification of benthic samples*; ZBD2016-11 *Quantifying benthic biodiversity across natural gradients*; ZBD2012-03 *Chatham Rise Benthos – Ocean Survey*; ZBD2014-10 *Benthic biodiversity.*
NZ government research (current)	MBIE programme: Sustainable Seas COIX1515 Sustainable Seas Ko Nga Moana Whakauka.
Related chapters/issues	Habitats of particular significance for fisheries management (HPSFM), marine environmental monitoring, marine mining/sand extraction, land-based effects.

Note: This chapter was fully updated for AEBAR 2014, since then it has been updated for accuracy only (statistics and obvious new developments only).

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 seines. 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 and Tasman Bays, 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, 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, all mapped as at 2015). 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 34 marine reserves protect about 7.6% of New Zealand's Territorial Sea; however, 99% 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 include 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 (30%) 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: 28% of underwater topographic features (a term that includes underwater hills, knolls, and seamounts); 52% of seamounts over 1000 m high; and 88% of known active hydrothermal vents.

11.2 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 20 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 in the past 10 years 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, NMFS 2002, NRC 2002, Dayton et al. 2002, Thrush & Dayton 2002, Lokkeborg 2005, Barnes & Thomas 2005, Clark & Koslow 2007).

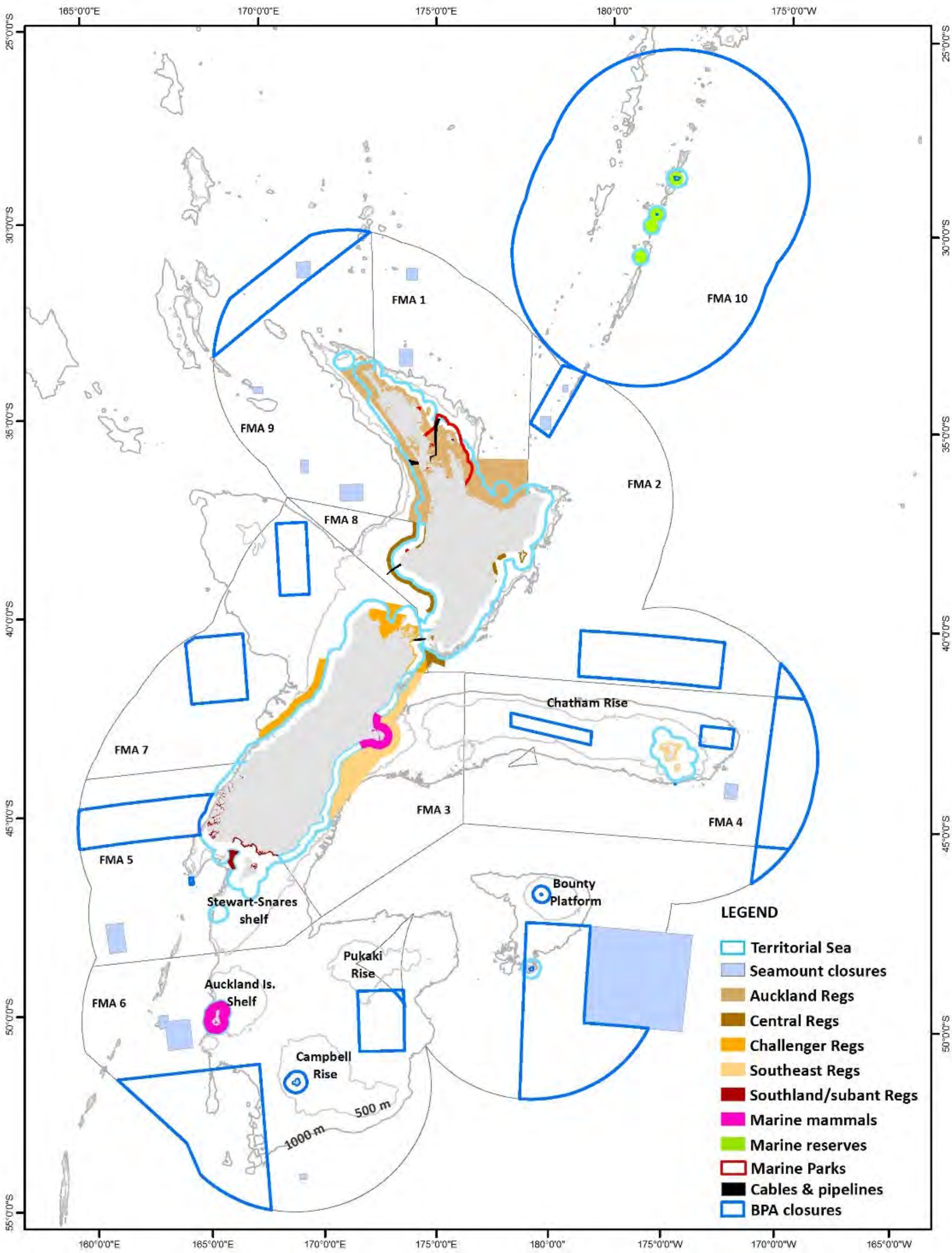


Figure 11.1: Map, adapted from Baird et al. 2011, of the major spatial restrictions to trawling and the Ministry for Primary Industries Fishery Management Areas (FMAs) within the outer boundary of the New Zealand EEZ. Vessels longer than 28 m may not trawl within the TS and additional restrictions are 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).

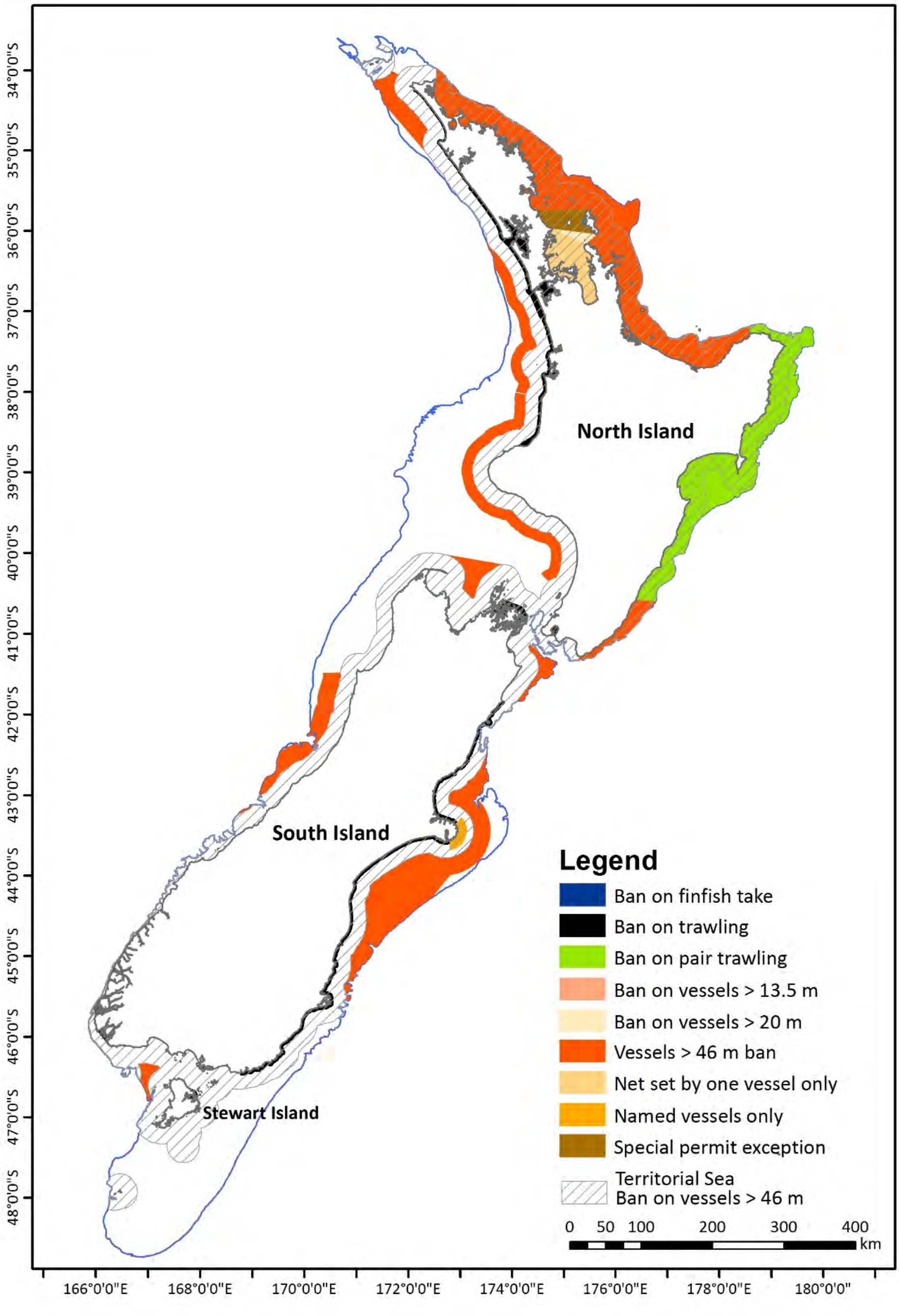


Figure 11.2: Areas showing where trawling is prohibited and other relevant restrictions apply in waters shallower than 250 m depth. Note the area shown as 'Ban on pair trawling' also is closed to vessels over 46 m (Baird et al. 2015).

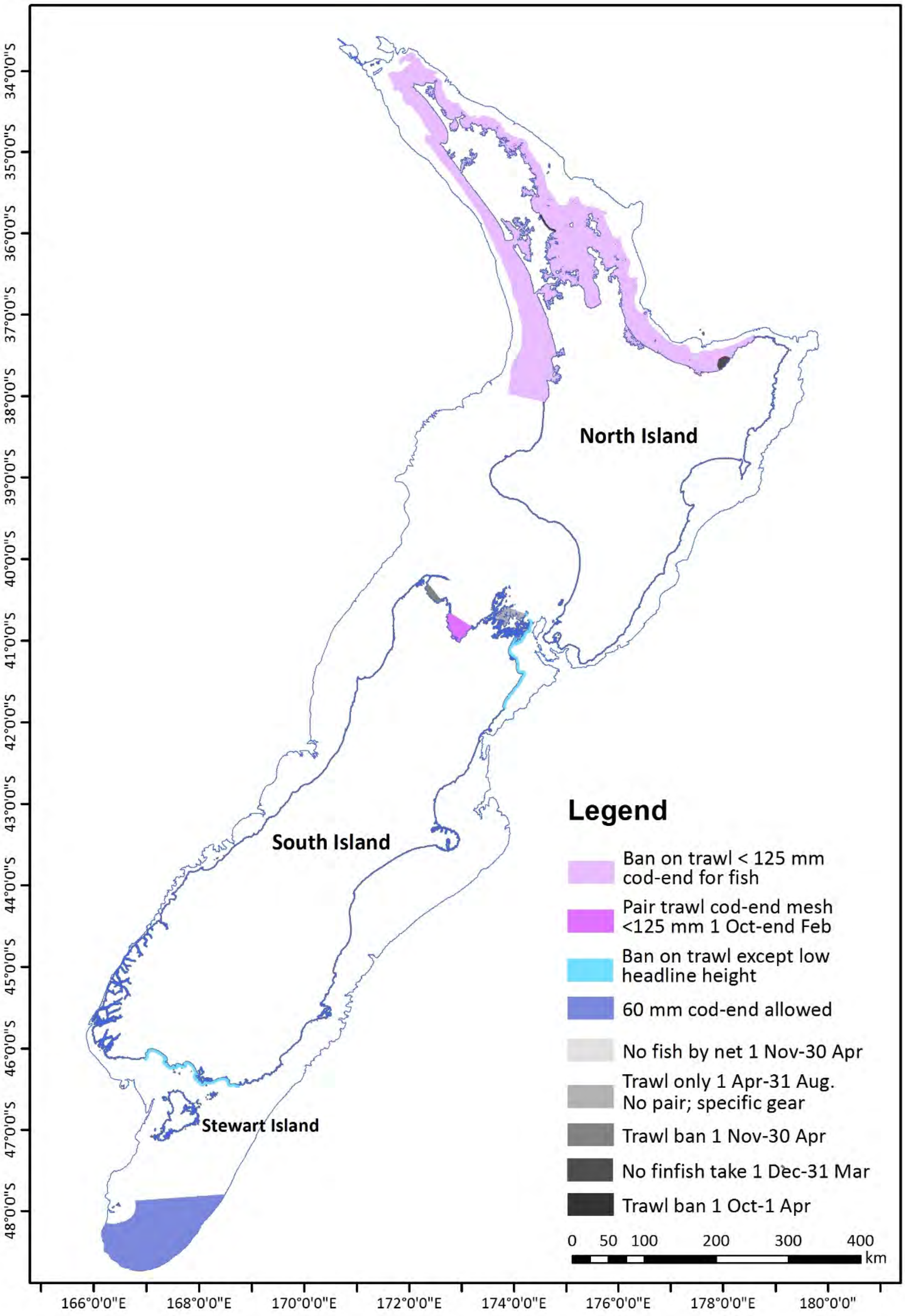


Figure 11.3: Areas where gear and seasonal restrictions apply to the use of trawl gear, in waters shallower than 250 m depth (Baird et al. 2015).

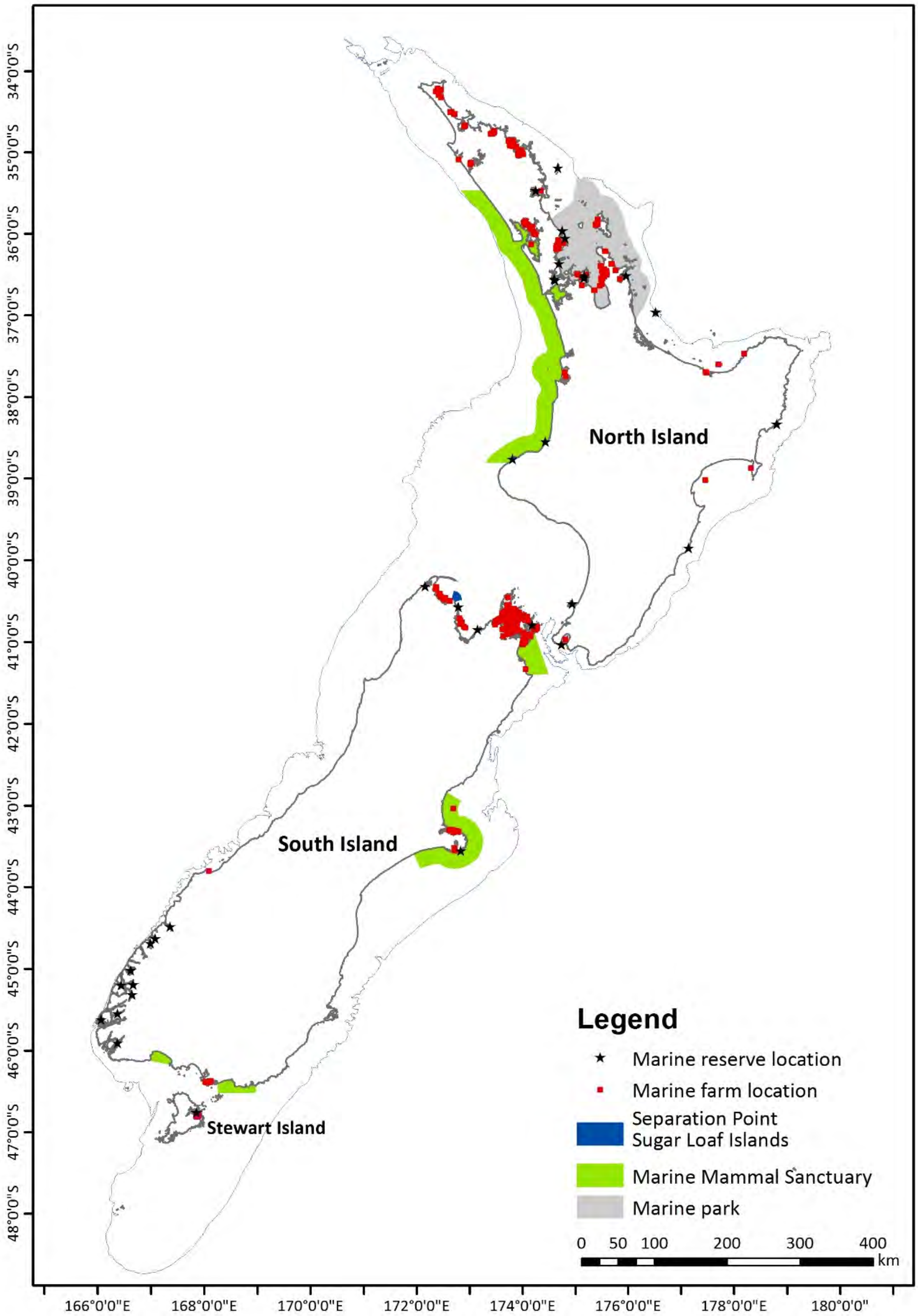


Figure 11.4: Points indicative of locations of marine reserves and marine farms, Separation Point and Sugar Loaf Islands closed areas, marine mammal sanctuaries, and marine parks, in waters shallower than 250 m depth (Baird et al. 2015).

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 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 in the sparse but highly diverse communities of the deep sea has developed only recently. The reviews above broadly indicate that numerical abundance of many invertebrate declines (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 resorting, 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-contact 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).

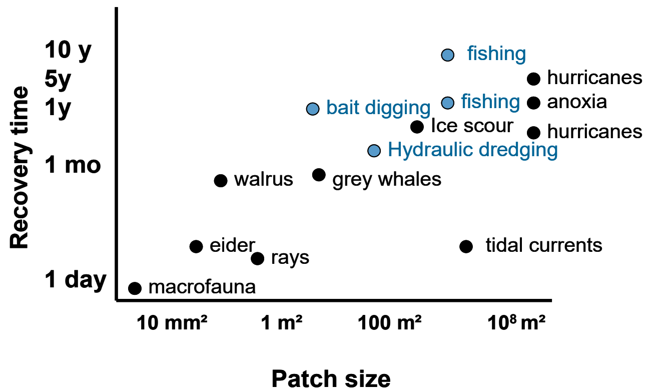


Figure 11.5: General relation between the spatial extent of disturbance events and the time taken to recover from such events in marine systems (after Kaiser et al. 2003). Blue dots signal human impacts, including fishing in habitats of different abilities to recover, and black dots signal natural disturbance.

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 NMFS panel);
  - The type of gear used and methods of deployment (all reviews, moderate to strong evidence 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. However, the positive or negative consequences for ecosystem processes such as production had not been addressed until more recently (e.g., Jennings et al. 2001a, 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. (2001a) 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.

11.3 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 will be dealt with in turn.

11.3.1 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. This 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 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, Table 11.1) 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.

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, see also: https://www.mfe.govt.nz/environmental-reporting/marine/fishing-activity-indicator/fishing-activity-seabed-trawling.html).

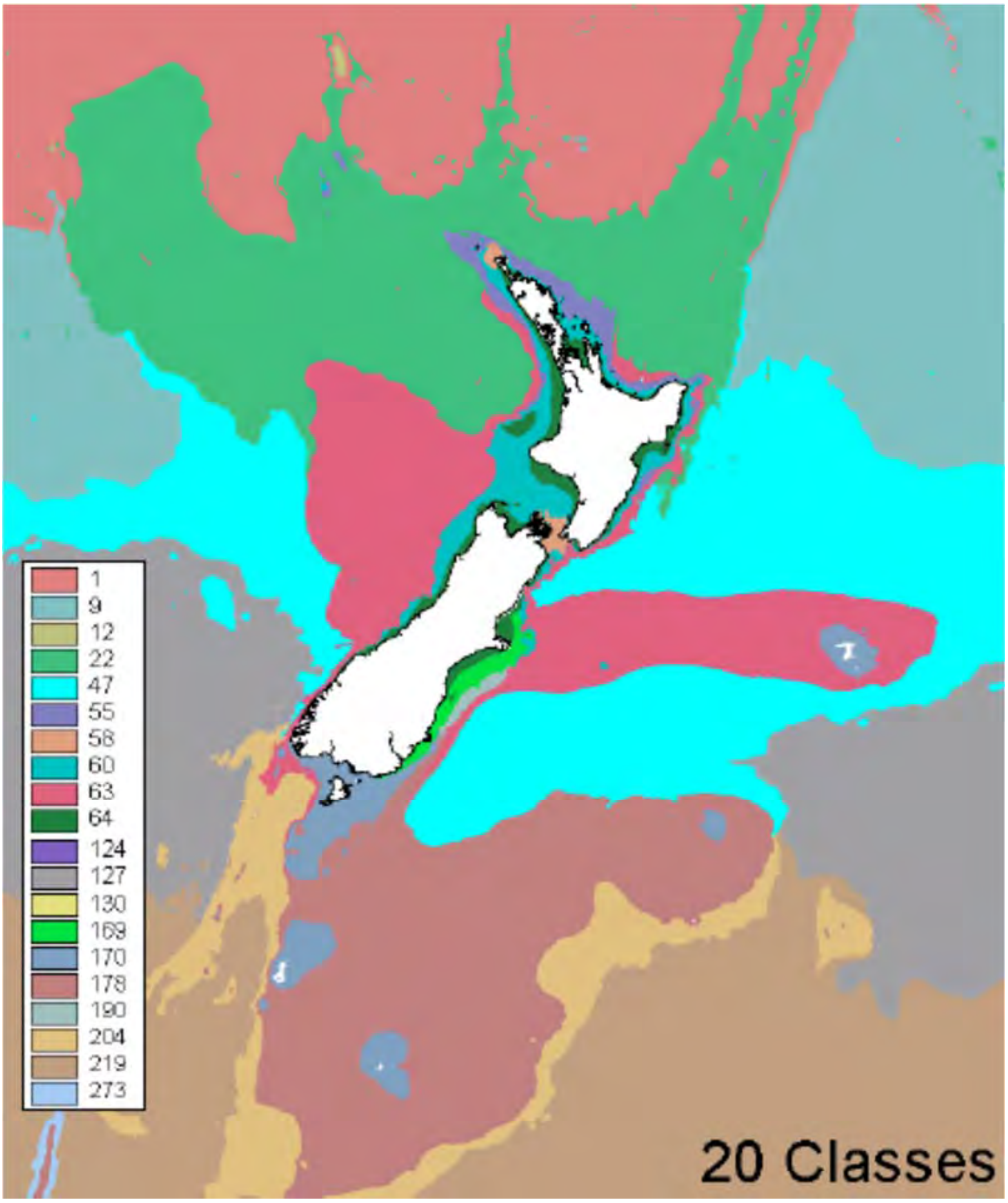


Figure 11.6: The 20-class version of the 2005 general purpose Marine Environment Classification (MEC, from Snelder et al. 2005). The class numbers are nominal; for attributes of each class at this level, see Table 11.1.

Based partly on this experience, the Ministry of Fisheries commissioned a Benthic-Optimised Marine Environment Classification, BOMEC. 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).

Recent testing (Bowden et al. 2011) has indicated that the BOMEC out-performs the original MEC at predicting benthic habitat classes on and around the Chatham Rise, but that none of the available classifications is 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.

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 like 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., Rowden et al. 2008), 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. 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. NIWA has a MBIE-funded project 'Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific region' in collaboration with Victoria University of Wellington and the Marine Conservation Institute (USA). The research will develop a model to predict the locations of VMEs to inform New Zealand and South Pacific Regional Fisheries Management Organisation (SPRFMO) initiatives on spatial management in the South Pacific region. There may be applications within the New Zealand EEZ.

Table 11.1: Average values for each of the eight defining environmental variables in each class of the 20-class level of the MEC classification. After Snelder et al. (2005).

Class	Area (km^2)	Depth	Slope	Orbital velocity	Radiation mean	SST amplitude	SST gradient	SST winter	Tidal current	2-class level	4-class level	9-class level
1	88,503	-3001	1.4	0	17.5	2.3	0.01	19.5	0.06	Oceanic	Subtropical	Deep
22	53,368	-1879	1.5	0	15.4	2.4	0.01	16.3	0.11			
9	64,306	-5345	1.4	0	14.8	2.6	0.01	16.1	0.03			Abyssal
47	60,053	-2998	1.0	0	12.1	2.4	0.01	11.6	0.07		Shelf and subtropical front	Central
55	2,213	-334	1.6	0	15.5	2.4	0.02	15.1	0.20			
63	26,626	-754	0.9	0	12.8	2.4	0.02	12.1	0.18			
178	39,360	-750	0.4	0	9.5	1.3	0.01	7.6	0.15			Southern
127	60,884	-4830	0.5	0	10.7	1.7	0.01	10.0	0.05		Sub-Antarctic	
204	18,277	-2044	3.0	0	9.2	0.9	0.01	8.0	0.08			
273	805	-2550	9.1	0	8.4	1.4	0.03	4.4	0.05			
219	93,982	-4779	0.6	0	8.9	1.0	0.01	6.7	0.04			
12	149	-94	0.9	113	17.8	2.3	0.01	19.3	0.30	Coastal		Northern
58	394	-117	0.7	57	14.7	2.2	0.03	13.0	1.09			Central
60	4,084	-112	0.3	21	14.4	2.5	0.02	13.2	0.26			
64	2,689	-38	0.3	272	14.2	2.9	0.02	12.6	0.19			
124	68	-8	0.4	836	13.4	2.3	0.02	12.7	0.00			
130	14	-10	0.4	353	14.1	2.4	0.09	11.9	0.21			
169	932	-66	0.2	113	12.4	2.7	0.04	9.9	0.21			
190	339	-321	1.9	3	12.3	2.3	0.06	9.4	0.10			
170	5,208	-129	0.3	99	10.2	1.3	0.02	9.3	0.55			Southern

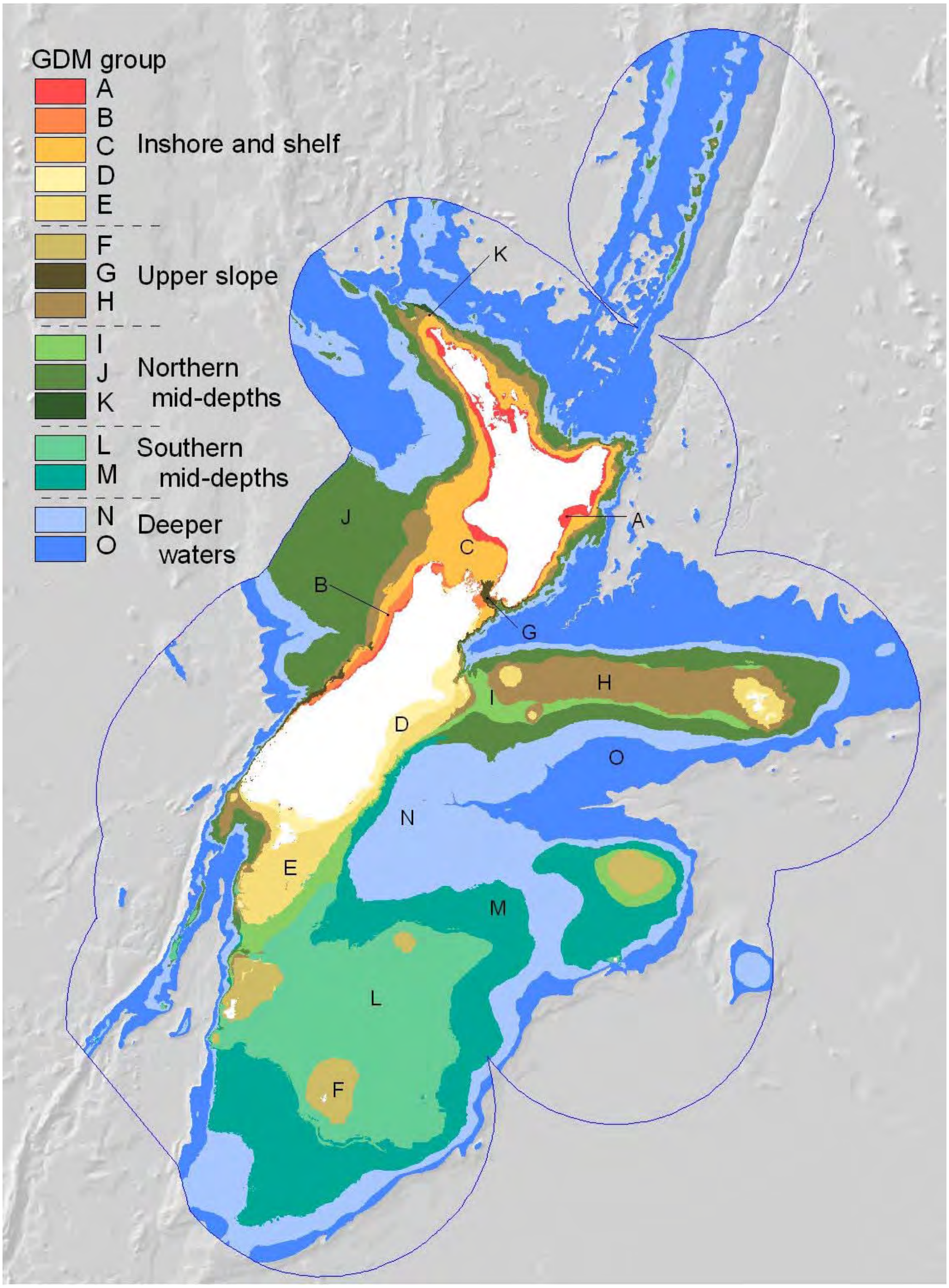


Figure 11.7: Map of the distribution of Benthic-Optimised Marine Environment Classification (BOMEC) classes defined by multivariate classification of environmental data transformed using results from GDM analyses of relationships between environment and species turnover averaged across eight taxonomic groups of benthic species. From Leathwick et al. (2012).

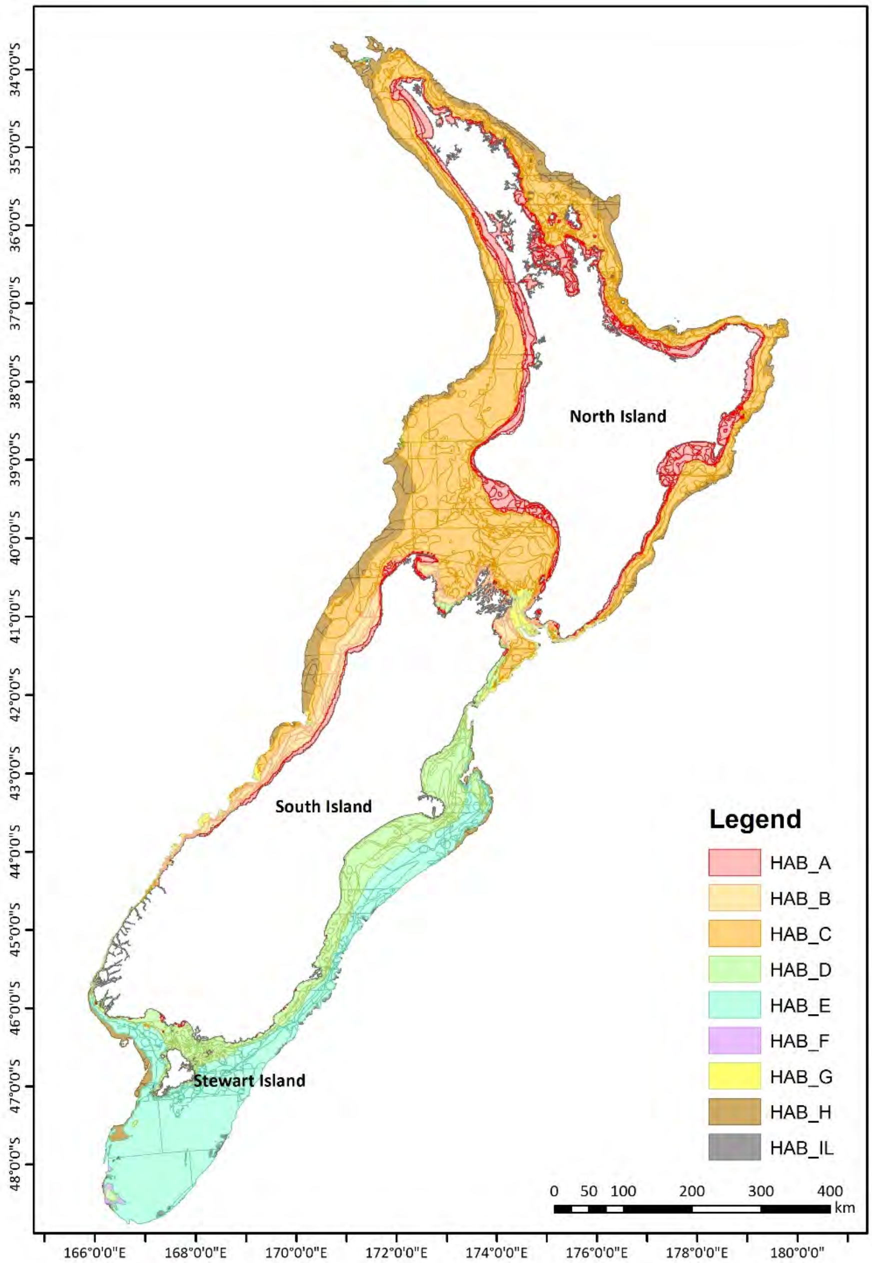


Figure 11.8: The broad habitat definitions based on the BOMEC classes, with divisions indicating areas of different sediment, depth zone, and statistical area in waters shallower than 250 m depth. From Baird et al. (2015).

11.3.2 DISTRIBUTION OF FISHING

Since 1989–90, mobile bottom fishing has been reported on one of three standardised reporting forms (Table 11.2). Trawl Catch Effort and Processing Returns (TCEPRs) contain detailed spatial and other information for each trawl tow, whereas Catch Effort and Landing Returns (CELRs) include only summarised information for each day's fishing, with very limited spatial resolution. 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. Between 1989–90 and 2004–05, only about 25% of all mobile bottom fishing events were reported on TCEPRs. Another 25% were bottom trawls reported on CELRs, and the remaining 50% were dredge tows for shellfish reported on CELRs. The distribution of trawling reported on CELRs is not the same as that reported on TCEPRs; the smaller trawlers using CELRs were much more likely than the larger boats to fish close to the coast and target inshore species such as flatfish, red cod, tarakihi, and red gurnard (collectively 73% of all trawl tows reported on CELRs).

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) and Catch Effort and Landing Returns (CELRs).

	Trawl catch and effort reporting forms		
	TCEPR	TCER	CELR
Year of introduction	1988–89	2007–08	1988–89
Vessels using	All trawlers >28 m Other vessels as directed Other vessels optional	All trawlers 6–28 m unless exempted	Trawlers not using TCER or TCEPR Shellfish dredgers
Trawl tow reporting	Tow by tow, start and finish locations, speed, depth, gear, duration	Tow by tow, start location, speed, depth, gear, duration	Daily summary, number of tows, gear, fishery area
Spatial resolution	1 minute (lat/long)	1 minute (lat/long)	Statistical reporting area (optionally lat/long)

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 mid-water trawl in contact with the bottom) in New Zealand's TS and EEZ during the 1990s and up to 2004–05, respectively, and this work has recently been updated to 2009–10 by Black et al. (2013) for data reported on TCEPR. They 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 and vice versa. 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, many inshore vessels used CELRs and these comprised a substantial proportion of reported trawling, even 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. Since 2007–08, almost all trawling effort has been reported on TCEPR or TCER forms.

Baird & Wood (2018) updated the three annual measures of fishing effort: the number of tows, the aggregate swept area (using assumed door spreads, see Figure 11.9), 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). Baird & Wood (2018) produced maps of the footprint, for each of the deepwater Tier 1 target species or species groups and for the Tier 1 and Tier 2 target species combined, summary maps of the aggregated swept area, and various tables and figures describing trends. The annual number of trawls peaked in 1997–98 at 57 195 tows (aggregate swept area about 166 092 km^2). In 2015–16, 25 572 tows were reported on TCEPRs (about 78 392 km^2).

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 like Figure 11.9, yet can represent intense fishing effort on particular, small seamount features (e.g., Rowden et al. 2005, O'Driscoll & Clark 2005).

Previous trawl footprint analyses (Baird et al. 2011, Black et al. 2013) have recognised that they underestimated trawl effort in inshore areas through exclusion of data recorded 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 (Figure 11.9, Figure 11.10).

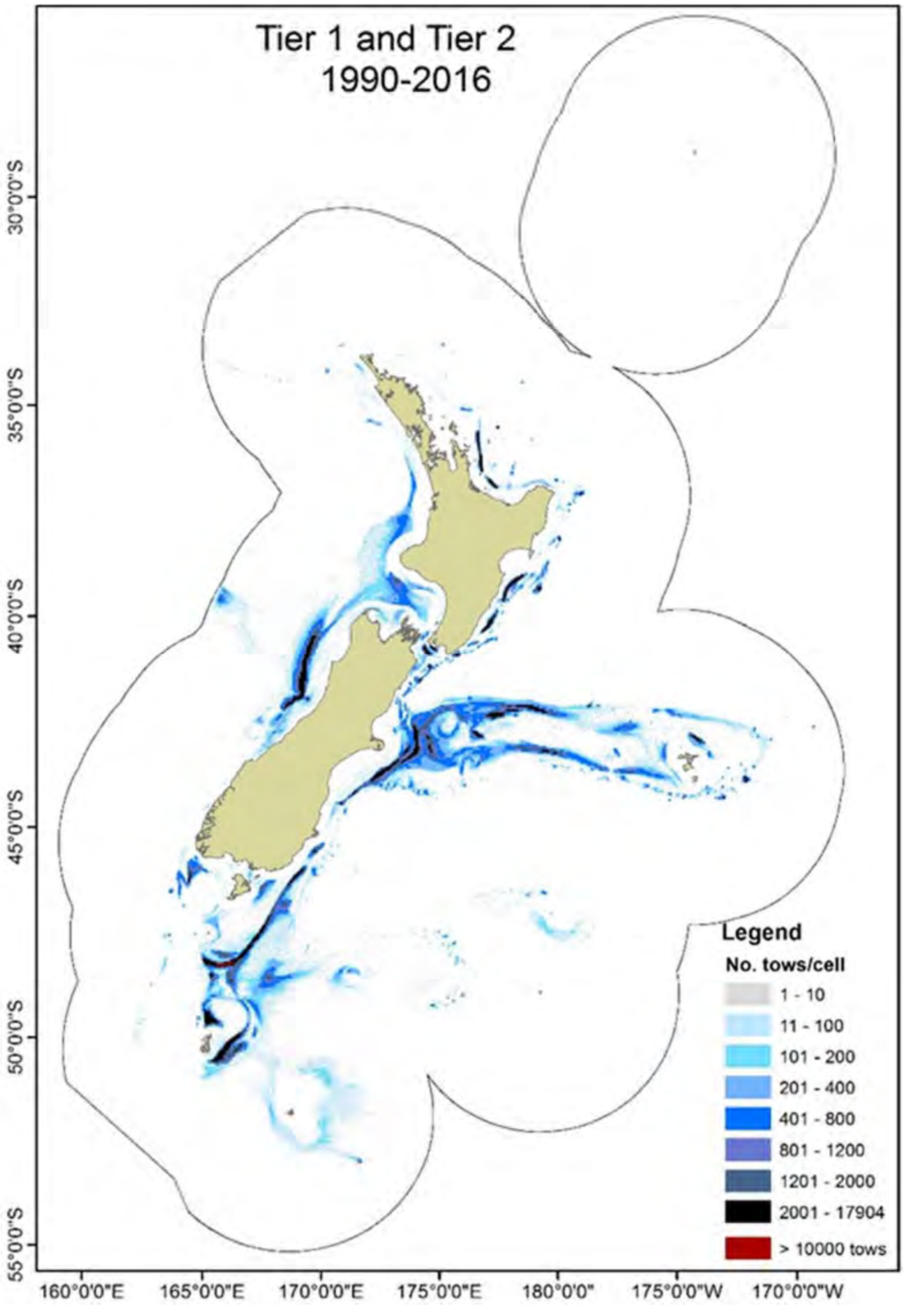


Figure 11.9: Map from Baird & Wood (2018) showing the frequency of bottom-contacting trawling effort reported on TCER and TCEPR forms 1989–90 to 2015–16. The colour scale indicates the frequency of bottom-contact trawling estimated by Baird & Wood for each 5 × 5 km cell, all deepwater Tier 1 and Tier 2 target fishstocks combined (i.e., the most frequently fished 25 km^2 cells had over 17 000 tows recorded over 27 years).

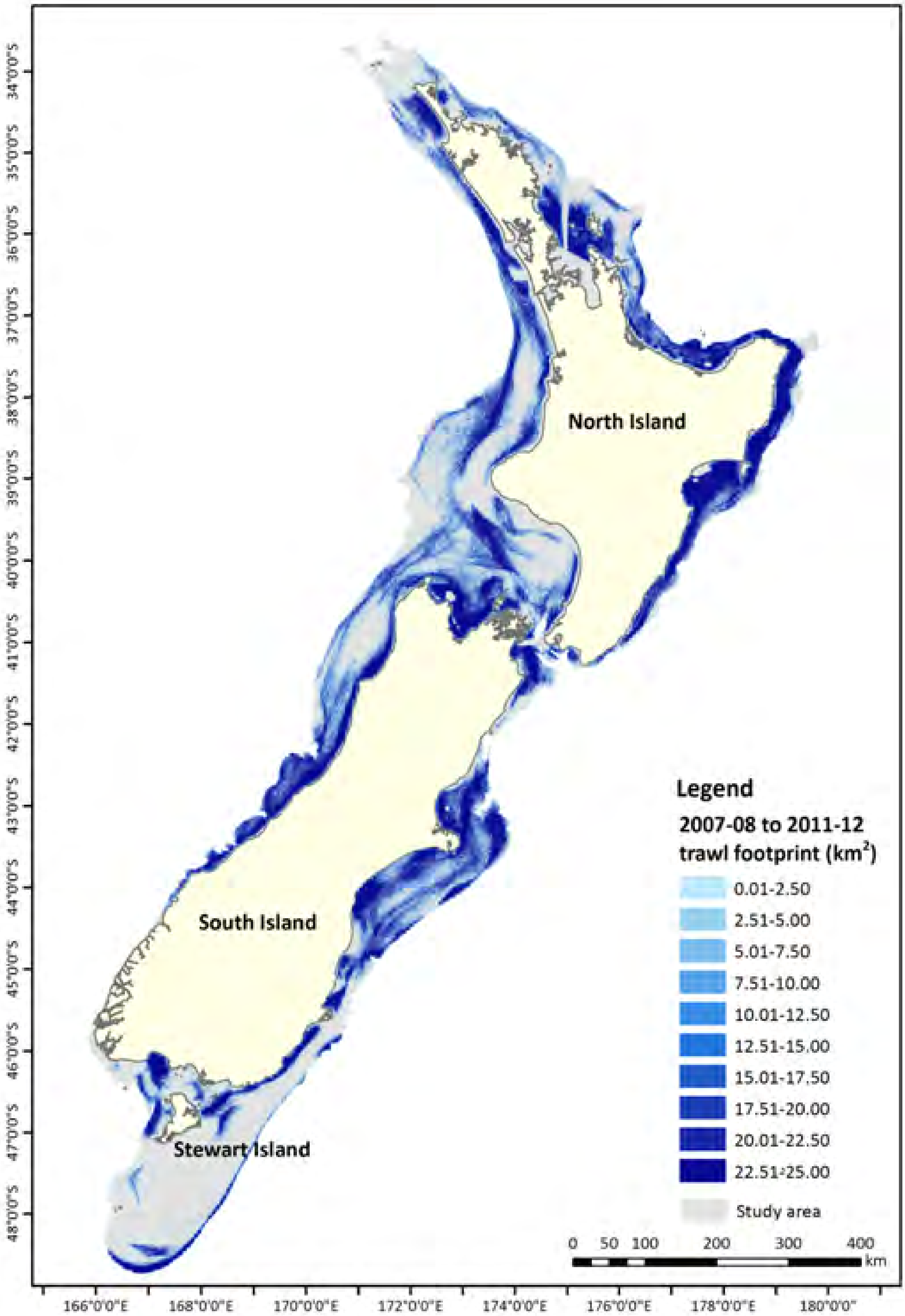


Figure 11.10: Total trawl cell based for footprint for the area shallower than 250 m for 2007–08 to 2011–12 combined (Baird et al. 2015).

After the peak of over 140 000 reported trawl tows in 1996–97 and 1997–98 (Figure 11.11) when slightly over half of all tows were reported on TCEPRs, overall trawling effort declined to less than 80 000 tows per year by 2013–14, only about 40% of which is reported on TCEPRs (virtually all other tows are reported on TCERs).

Dredging for shellfish (oysters and scallops) is conducted in a number of specific areas that have separate, smaller statistical reporting areas (Figure 11.12). Over the 16-year dataset, there were almost 1.5 million scallop dredge tows in the four main scallop fisheries and over 0.6 million oyster dredge tows in the two dredge oyster fisheries. These data are collected on CELRs, usually at the spatial scale of a scallop or oyster fishery area and the data have been 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 (Figure 11.13).

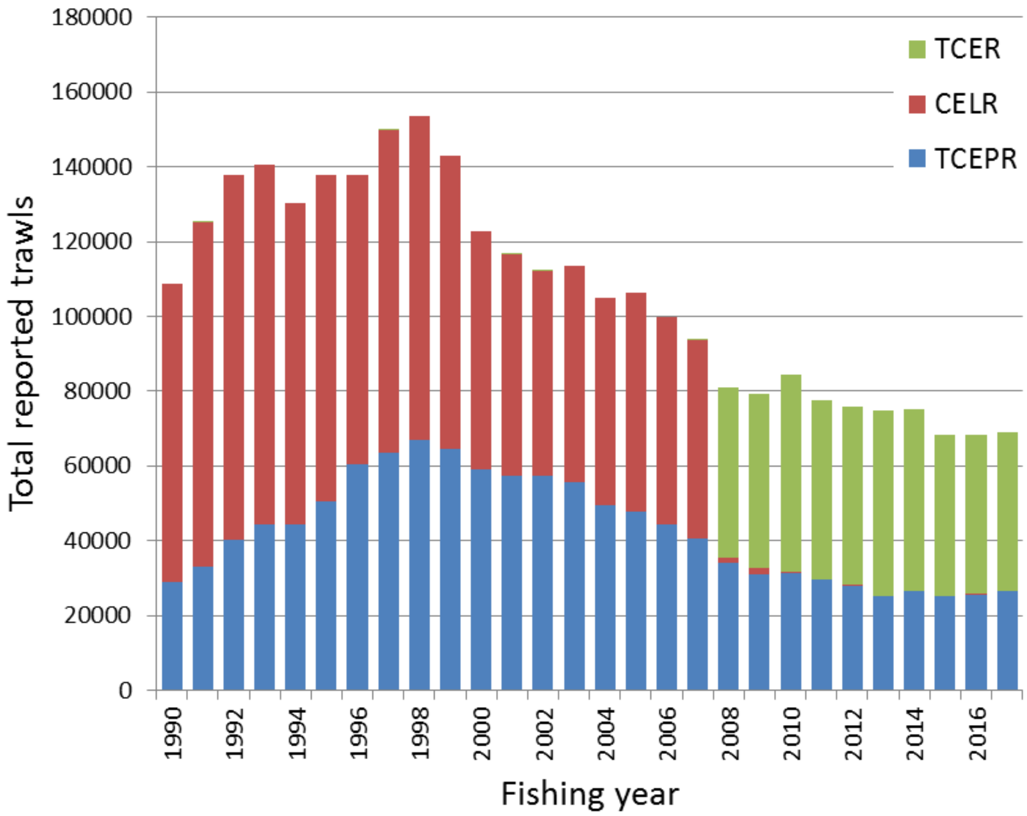


Figure 11.11: The number of trawl tows reported on Trawl Catch Effort and Processing Returns (TCEPR), Catch Effort and Landing Returns (CELR) and Trawl Catch and Effort Return (TCER) between the 1989–90 (1990) and 2016–17 (2017) fishing years. Data for the 2016–17 year may be incomplete.

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 is not reported with such high precision, but records kept by fishers in industry logbooks are often much more detailed than the Ministry for Primary Industries standard returns, and have sometimes been used to support spatial analyses that would not have been possible using the standard 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). These studies indicate the value of records with higher spatial precision.

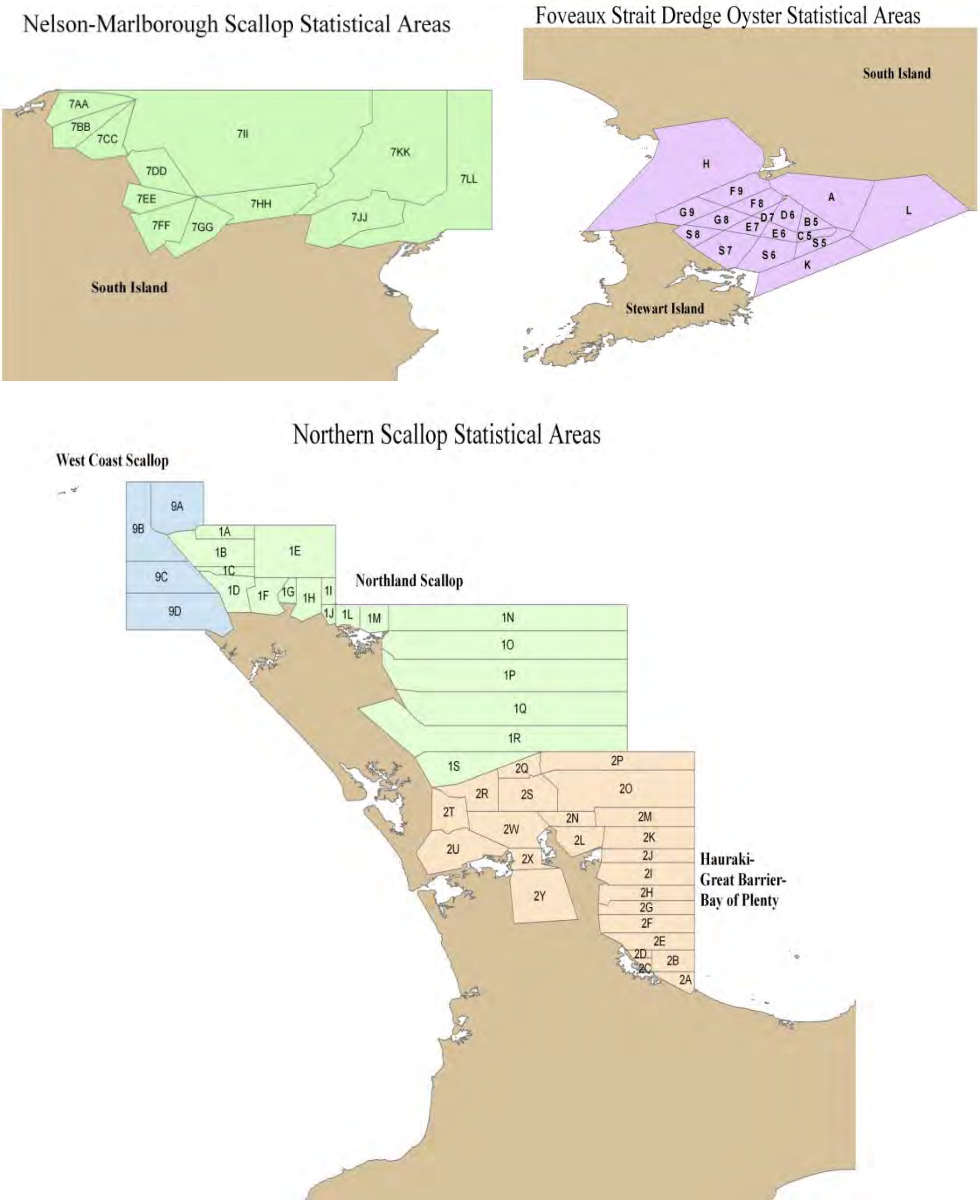


Figure 11.12: Maps taken from Baird et al. (2011) of statistical reporting areas for the main oyster and scallop dredge fisheries (scales differ). Note that these reporting areas are generally much smaller than the standard statistical reporting areas used for most finfish reporting.

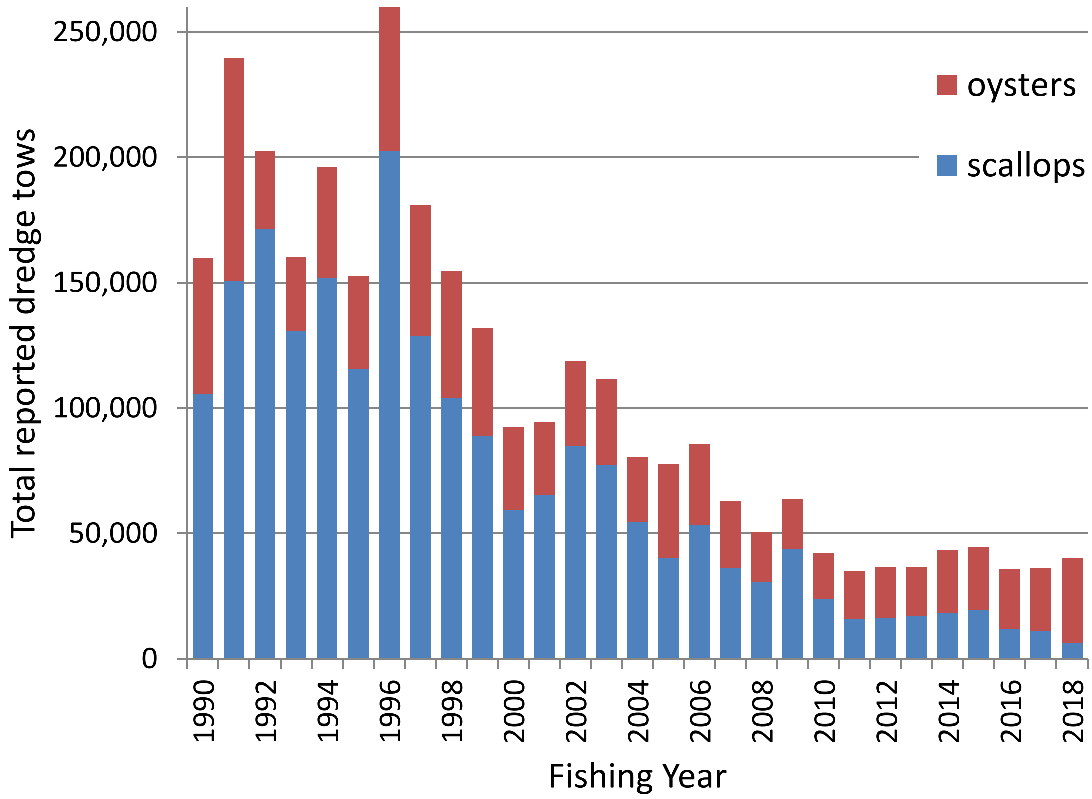


Figure 11.13: The number of dredge tows for scallop or oysters reported on Catch Effort and Landing Returns (CELR) between the 1989–90 (1990) and 2017–18 (2018) fishing years (data from Baird et al. 2011 and MPI databases). Data for the 2017–18 year will be incomplete.

11.3.3 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 (2018) overlaid the 1989–90 to 2015–16 fishing year deepwater Tier 1 and Tier 2 fishstock footprint on the 15-class BOMEC to estimate the proportion of each class that had been trawled (and reported on TCERs and TCEPRs) (Figure 11.14). They found that the size of the footprint and the proportion of each class trawled varied substantially between habitat classes (Table 11.3, Figure 11.15). Class O is the largest BOMEC class but has almost no reported fishing effort; this class is mainly beyond trawlable depths. Conversely, class I is one of the smaller classes but has a larger trawl footprint that overlays 74% of the total class area. Two contrasting classes, together with their trawl footprints, are shown in Figure 11.16, based on analysis up to 2015–16. The trawl footprint from the Baird & Wood (2018) analysis overlaps about 13% of the 2.6 million km^2 of seafloor covered by the BOMEC, about 8% of the 4.1 million km^2 of seafloor within the New Zealand EEZ boundary (i.e., including the Territorial Sea), or about 23% of the area open to trawling in depths down to 1600 m. However, these overlays and that for some individual BOMEC classes (particularly coastal classes A–E) do not represent the total bottom-contacting trawl footprint because the analysis was restricted to trawl effort for deepwater Tier 1 and Tier 2 target fishstocks, and CELR data during 1990–2007 were omitted. For example, this trawl footprint included only the ‘deepwater’ fishstocks for species such as barracouta and ling, and inshore species such as snapper and tarakihi were not included.

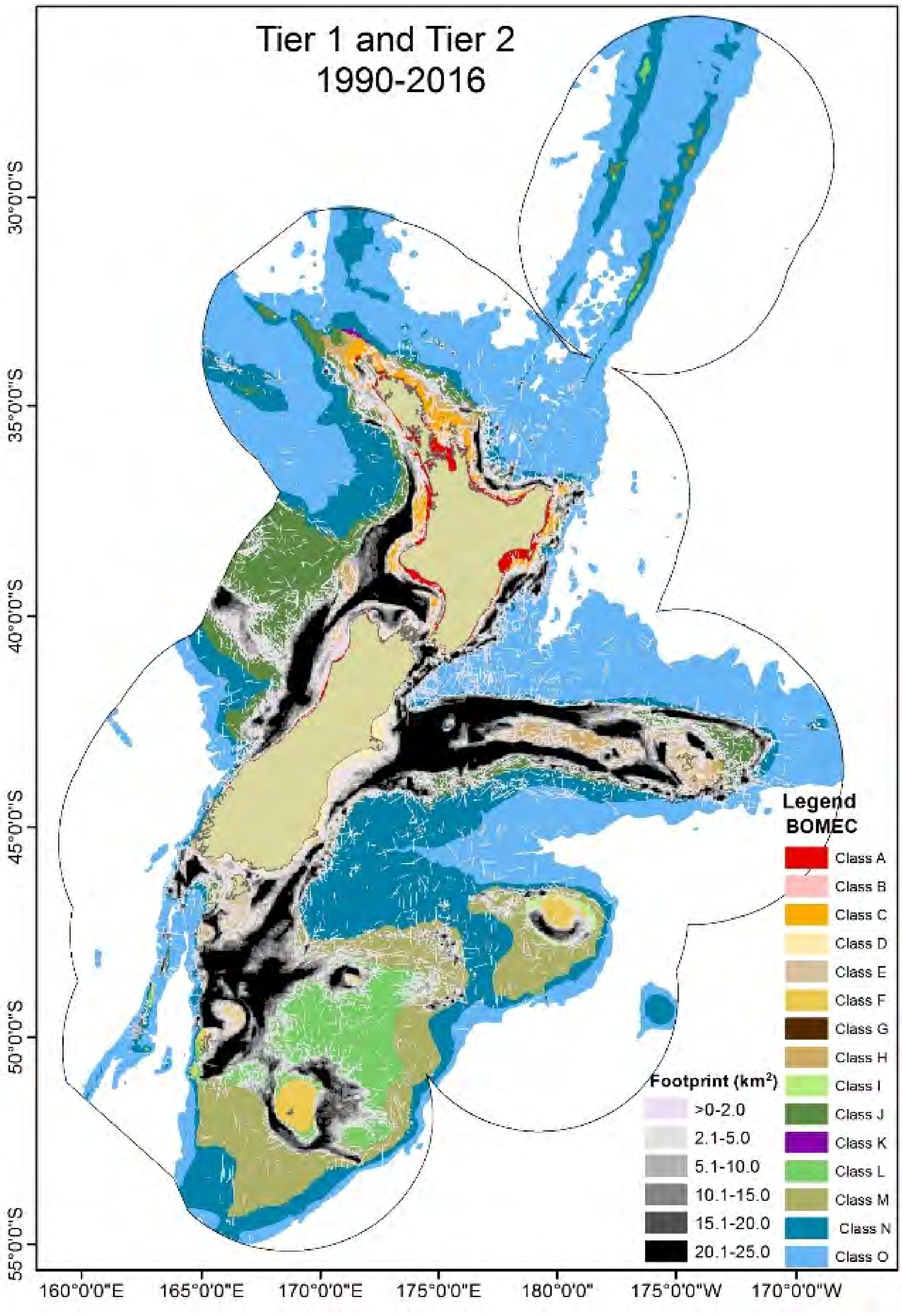


Figure 11.14: Plot from Baird and Wood (2018) of the distribution of the 1990–2016 trawl footprint for deepwater Tier 1 and Tier 2 target fishstocks, relative to the 15 BOMEC classes (to depths of 3000 m) (after Leathwick et al. 2012).

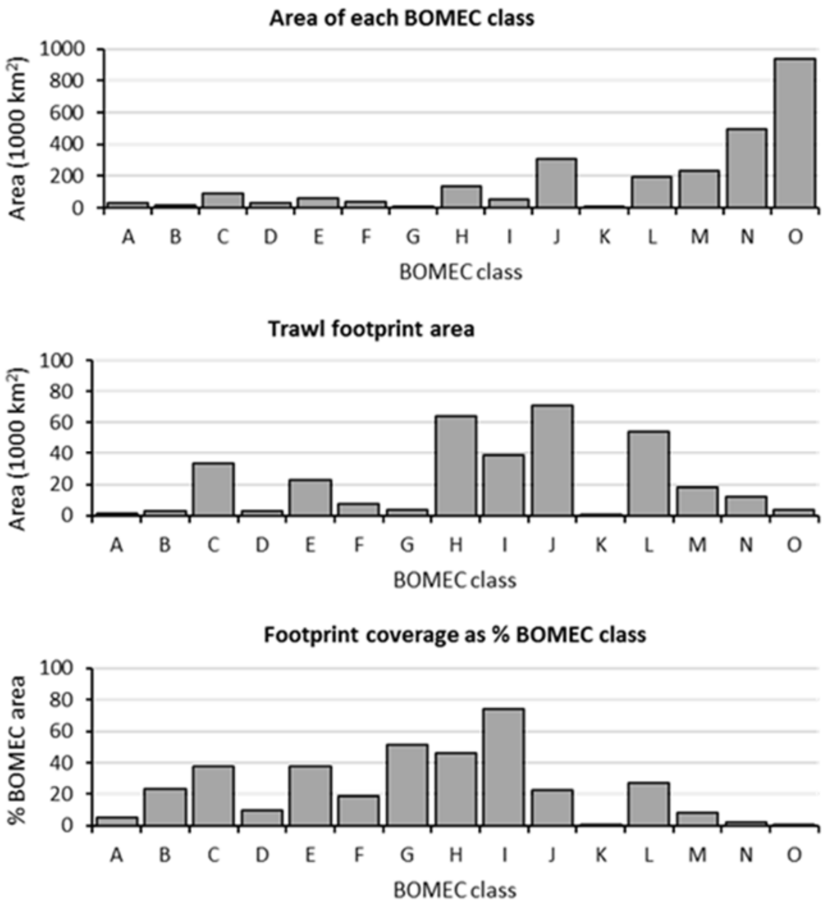


Figure 11.15: Plots from data provided by Baird & Wood (2018) of the areas of each BOMEC class (top), the area of the fishing footprint up to 2015–16 (centre), as shown in Figure 11.9, and percentage of each BOMEC class area covered by the fishing footprint (bottom).

Table 11.3: Estimated area of each BOMEC class (within the outer boundary of the EEZ) and fishing footprint from TCER and TCEPR deepwater Tier 1 and Tier 2 target fishstocks over the fishing years 1989–90 to 2015–16 (Baird & Wood 2018).

BOMEC class	Area (km^2)	Footprint area (km^2)	Footprint area (%)
A*	27 557	1 421.7	5.2%
B*	12 420	2 889.9	23.3%
C*	89 710	33 527.2	37.4%
D*	27 268	2 676.9	9.8%
E*	60 990	22 792.3	37.4%
F	38 608	7 114.4	18.4%
G	6 342	3 267.1	51.5%
H	138 550	63 918.0	46.1%
I	52 224	38 716.3	74.1%
J	311 361	70 784.1	22.7%
K	1 290	3.2	0.2%
L	198 577	54 179.7	27.3%
M	233 825	18 220.5	7.8%
N	493 034	11 715.5	2.4%
O	935 315	3 344.6	0.4%
Total	2 627 073	334 571.5	12.7%

* The trawl footprint and proportion overlapped in coastal classes A–E will be grossly underestimated because CELR data are excluded. A comparison with the 2016 version of this table in the AEBAR shows a decrease in the latest report of the

percentage footprint coverage for some areas and in total. This appears due to the faulty inclusion of some inshore stocks in this table in the 2016 AEBAR.

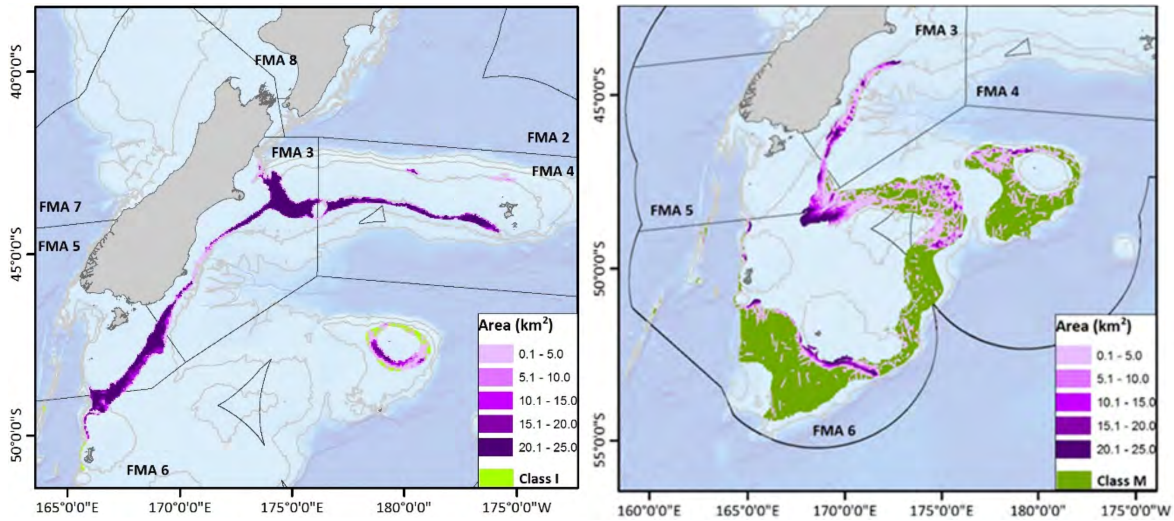


Figure 11.16: Maps created from Baird & Wood (2018) showing BOMEC classes I (left) and M (right) overlaid with the footprint of deepwater Tier 1 and Tier 2 fishstock trawls on or near the seafloor reported on TCER and TCEPR forms to 1990–16 for each 25-km^2 cell. Grey contour lines indicate depths of 500 m, 1000 m and 1500 m.

Baird et al. (2015) overlaid the combined TCER and TCEPR 2007–08 to 2011–12 fishing years trawl footprint on a classification for benthic habitats shallower than 250 m (presented in Figure 18 of that report). As with the offshore data, the size of the footprint and proportion of each class trawled varied between habitat classes (Table 11.4), and ranged from 21% (class F) to 76% (class B). Over the 2007–08 to 2011–12 period, the trawl footprint overlays 48% of the area shallower than 250 m.

Table 11.4: Areas* of the separate habitat classes and areas† of the five-year trawl footprint in each habitat class, for the BOMEC classes, depth zones, and sediment types, and the percentage of the five-year trawl footprint in each class (see Baird et al. 2015 for more details).

Habitat class descriptors	Habitat class area (km^2)	Total footprint (km^2)	% area with trawl contact
BOMEC class			
A	27 375.2	14 047.1	52.1
B	12 318.8	9 322.3	75.7
C	89 560.4	47 120.0	52.6
D	25 513.1	16 344.7	64.4
E	47 186.8	13 890.6	29.6
F	381.7	73.2	21.1
G	3 898.4	1 902.9	50.8
H	25 204.4	9 228.1	36.8
I	473.2	340.8	72.0
J	133.9	31.3	30.0
L	188.9	121.5	67.6
Depth zone			
< 50 m	50 781.3	29 529.4	58.8
50–100 m	63 493.9	37 375.2	59.1
100–250 m	117 959.8	45 517.9	38.7
Sediment type			
Sand	104 830.2	54 851.9	52.4

Mud	72 518.1	41 001.8	57.0
Gravel	18 530.0	9 339.3	50.6
Sandy mud	203.4	203.4	99.98
Calcareous sand	6 763.7	2 080.7	31.3
Calcareous gravel	29 389.5	4 945.4	17.0
All	232 235.0	112 422.5	48.4

* The area measures for the habitat classes include any seafloor closed to trawling.

† The area measures for the five-year footprint represent 98.8% of the total footprint.

11.3.4 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 depth, 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 11.5). 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 like 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 11.5: Summary of studies of the effects of bottom trawling and dredging in New Zealand waters. [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

Table 11.5 [Continued]:

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 and Golden Bays. Bottom trawl, scallop and oyster dredge	Observational, gradient analysis	A gradient analysis was adopted to investigate the importance of the different factors affecting epifaunal and infaunal communities in Tasman and Golden Bays. Fishing was consistently identified as an 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.	Tuck et al. 2011
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 et al. 2010a, 2010b Williams et al. 2011

11.3.5 CURRENT RESEARCH

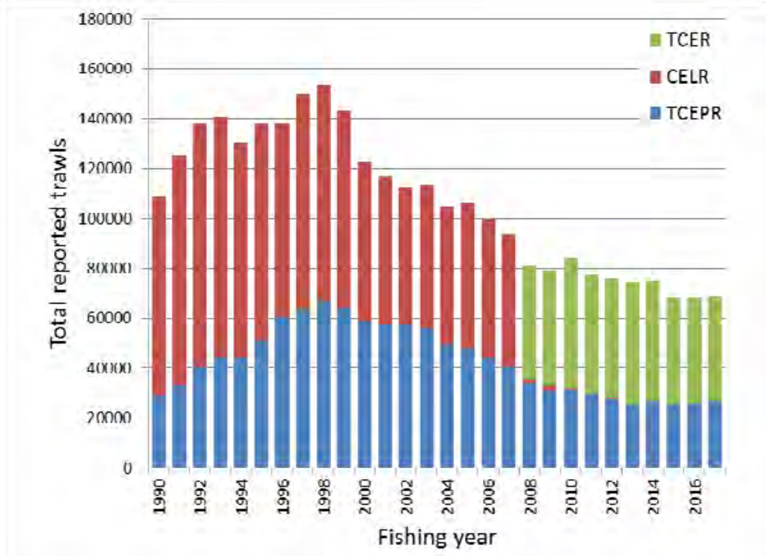
Project BEN2007/01 is a five-year project to assess the effects of fishing on soft sediment habitat, fauna, and processes across the range of habitat types in the TS and EEZ. Sampling and analytical strategies for such broad-scale assessments have been developed and the project has moved into a phase of data collection, collation, and analysis. Two field-based 'case studies' in different habitat types will be assessed, and a variety of existing information will be drawn together and analysed to provide a TS and EEZ-wide perspective. The focus of this study is on the relative sensitivities of different habitats in the TS and EEZ to disturbance by mobile bottom fishing methods.

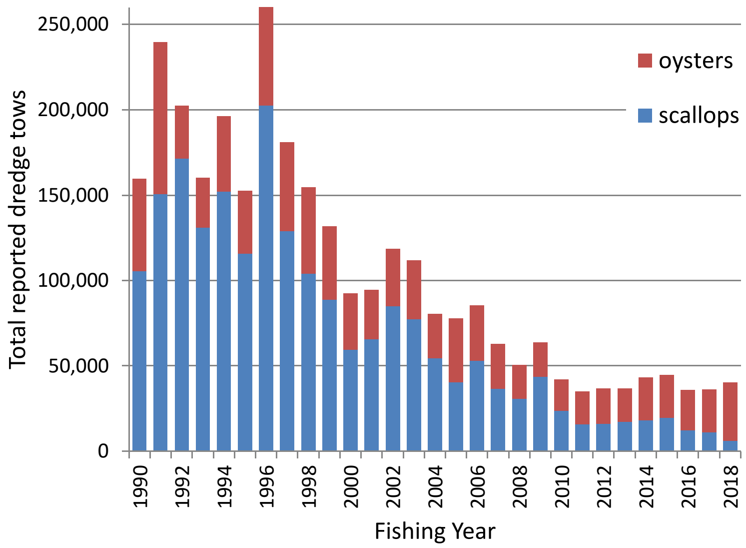
Project DAE2010/04 provides for an annual assessment of the 'footprint' of middle depth and deepwater trawl fisheries, including the overlap of the footprint with various depth ranges and habitat classes. Inshore fisheries, including shellfish dredge fisheries, are not covered under this project, so the focus is on offshore fisheries and habitats.

Project ZBD2012/03 will use data collected from recent Oceans Survey 20/20 sampling on the Chatham Rise to determine whether there are quantifiable effects of variations in seabed trawling intensity on benthic communities, and also conduct seabed mapping and photographic surveys in previously unsampled areas on the central crest of the rise.

Several MBIE-funded projects also have strong linkages with MPI research on benthic impacts. These include 'Vulnerable Deep-Sea Communities' (CO1X0906), which is analysing the time series of data from the 'Graveyard seamounts' (surveys in 2001, 2006, 2009, all carried out with support from MFish or the cross-departmental Oceans Survey 20/20 programme), as well as evaluating 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.

11.4 INDICATORS AND TRENDS

Annual number of tows	2016–17 fishing year: 68 868 trawl tows 40 361 shellfish dredge tows
Trend in number of tows	Trawl and dredge effort stable or decreasing in recent years: 



Cumulative overlap of TCEPR trawl footprint with BOMEC habitat classes for 1989–90 to 2015–16

BOMEC class	Area (km^2)	Footprint area (km^2)	Footprint area (%)
A*	27 557	1 421.7	5.2%
B*	12 420	2 889.9	23.3%
C*	89 710	33 527.2	37.4%
D*	27 268	2 676.9	9.8%
E*	60 990	22 792.3	37.4%
F	38 608	7 114.4	18.4%
G	6 342	3 267.1	51.5%
H	138 550	63 918.0	46.1%
I	52 224	38 716.3	74.1%
J	311 361	70 784.1	22.7%
K	1 290	3.2	0.2%
L	198 577	54 179.7	27.3%
M	233 825	18 220.5	7.8%
N	493 034	11 715.5	2.4%
O	935 315	3 344.6	0.4%
Total	2 627 073	334 571.5	12.7%

Cumulative overlap of trawl footprint shallower than 250 m with BOMEC habitat classes, depth zones and sediment types for 2007–08 to 2011–12

Habitat class descriptors	Habitat class area (km²)	Total footprint (km²)	% area with trawl contact
BOMEC class			
A	27 375.2	14 047.1	52.1
B	12 318.8	9 322.3	75.7
C	89 560.4	47 120.0	52.6
D	25 513.1	16 344.7	64.4
E	47 186.8	13 890.6	29.6
F	381.7	73.2	21.1
G	3 898.4	1 902.9	50.8
H	25 204.4	9 228.1	36.8
I	473.2	340.8	72.0
J	133.9	31.3	30.0
L	188.9	121.5	67.6
Depth zone			
< 50 m	50 781.3	29 529.4	58.8
50–100 m	63 493.9	37 375.2	59.1
100–250 m	117 959.8	45 517.9	38.7
Sediment type			
Sand	104 830.2	54 851.9	52.4
Mud	72 518.1	41 001.8	57.0
Gravel	18 530.0	9 339.3	50.6
Sandy mud	203.4	203.4	99.98
Calcareous sand	6 763.7	2 080.7	31.3
Calcareous gravel	29 389.5	4 945.4	17.0
All	232 235.0	112 422.5	48.4

* The trawl footprint and proportion overlapped in coastal classes A–E will be grossly underestimated because CELR data are excluded.

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THEME 4: ECOSYSTEM EFFECTS

12 NEW ZEALAND'S CLIMATE AND OCEANIC SETTING

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 fisheries and the broader marine environment. • Allows for improved understanding of the links between observed patterns and drivers of biological processes. • Allows for testing of likely future scenarios • New Zealand trends of increasing air and sea temperatures and ocean acidification are consistent with global trends.
Emerging issues	• New Zealand's oceanic climate is changing. • Causal mechanisms that link the dynamics of a variable marine environment to variation in biological productivity, particularly of fisheries and biodiversity, are not well understood in New Zealand or internationally, but are the subject of multiple studies. • The 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 to be 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 is likely to impact fish productivity.
MPI research (current)	ZBD2013-02 *VME connectivity*; ZBD2014-01 *Live DW coral experiment phase 2*; ZBD2013-08 *NZ-Ross sea connectivity Humpback whales*; ZBD2014-10 *BPA biodiversity*; ZBD2014-03 *Sublethal effects of environment change on fish populations*; ZBD2014-04 *Isoscapes for trophic studies*; ZBD2014-05 *Ocean acidification*; ZBD2014-06 *Macroalgae mapping and potential as national scale indicators*; ZBD2014-07 *Southern coralline algae shellfish habitat*; ZBD2014-09 *Climate change risks and opportunities*; ZBD2013-03 *Continuous plankton recorder (16849)*; ZBD2013-06 *Shell generation*; ZBD2012-03 *CRise benthos (16589).*
Government and other research	NIWA Coast & Oceans Centre, 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; Statistics New Zealand Environmental Domain review; Department of Conservation the impacts of climate change on marine protected species. Relevant global climate programmes: Argo; Southern Ocean Observing System. Dragonfly science: Fast-forward fish: resilience of exploited marine populations to a changing ocean.
Related chapters/issues	This chapter provides background environmental information that is relevant to all chapters, but is of particular relevance to the following chapters: Biodiversity; Trophic and ecosystem-level effects.

Note: This chapter has not been updated since the AEBAR 2015.

12.1 CONTEXT

This chapter summarises information on ocean acidification, and oceanography around New Zealand, and climate trends in the Southwest Pacific region. This information provides context to understand the interaction 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 fishstocks and fisheries. The most recent analysis of trends in climate and oceanographic variables relevant to fisheries management in New Zealand is given in Hurst et al. (2012).

New Zealand is essentially part of a large submerged continent (Figure 12.1). The Territorial Sea (TS, extending from mean low water shore line to 12 nautical miles), Exclusive Economic Zone (the EEZ, extending from 12 nautical miles to 200 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 30°S in warm subtropical waters to 56°S in cooler, subantarctic waters, and 30 degrees of longitude from 161°E in the Tasman Sea to 171°W in the west Pacific Ocean. New Zealand's coastline, with its numerous embayments, is also long, with estimates ranging from 15 000 to 18 000 km, depending on the method used for measurement (Gordon et al. 2010).

New Zealand lies across an active subduction zone in the western Pacific plate; tectonic activity and volcanism have resulted in a diverse and varied seascape 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, the Challenger Plateau, the Campbell Plateau, and the Chatham Rise.

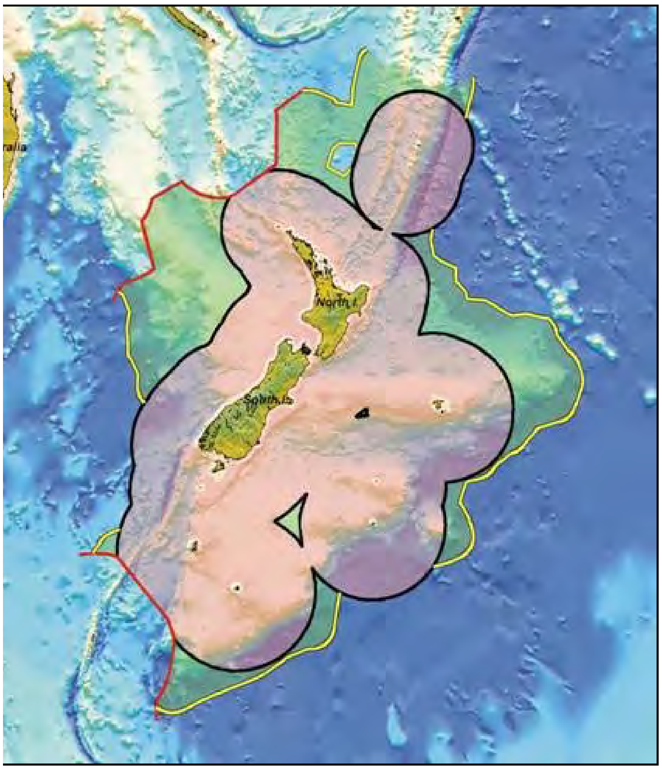


Figure 12.1: New Zealand land mass area 250 000 km^2; EEZ and territorial sea area (pink) 4 200 000 km^2; extended continental shelf extension area (light green) 1 700 000 km^2; Total area of marine jurisdiction 5 900 000 km^2. The black line shows the boundary of the New Zealand EEZ, the yellow line indicates the extension to New Zealand's legal continental shelf, and the red line the agreed Australia/New Zealand boundary under UNCLOS Article 76. Image courtesy of GNS.

The physical oceanography of the deep seas around New Zealand has recently been reviewed by Chiswell et al. (2015). Multiple platforms including satellites, drifting and profiling floats, moorings and oceanographic cruises have provided a wealth of new observations over the last 30 years and analysis of these observations has substantially improved our understanding of the oceanography over the basic description of the circulation around New Zealand that was known in 1985. Chiswell et al. summarise and integrate earlier research through a series of schematics of the ocean currents around New Zealand. The surface currents are shown in Figure 12.2.

The Tasman Sea, to the west of New Zealand, is separated 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°S to 32°S (Ridgway & Dunn 2003). Part of the separated flow crosses the Tasman Sea as the Tasman Front (Stanton 1981, Ridgway & Dunn 2003, Sutton & Bowen

2014) while the remaining flow continues south in the EAC extension. Sutton & Bowen (2014) found that the Tasman Front is a weaker connection than previously thought between the EAC and East Auckland Current (EAUC). They found that the Tasman Front was shallower than both the EAC and EAUC and transported less water than the other currents.

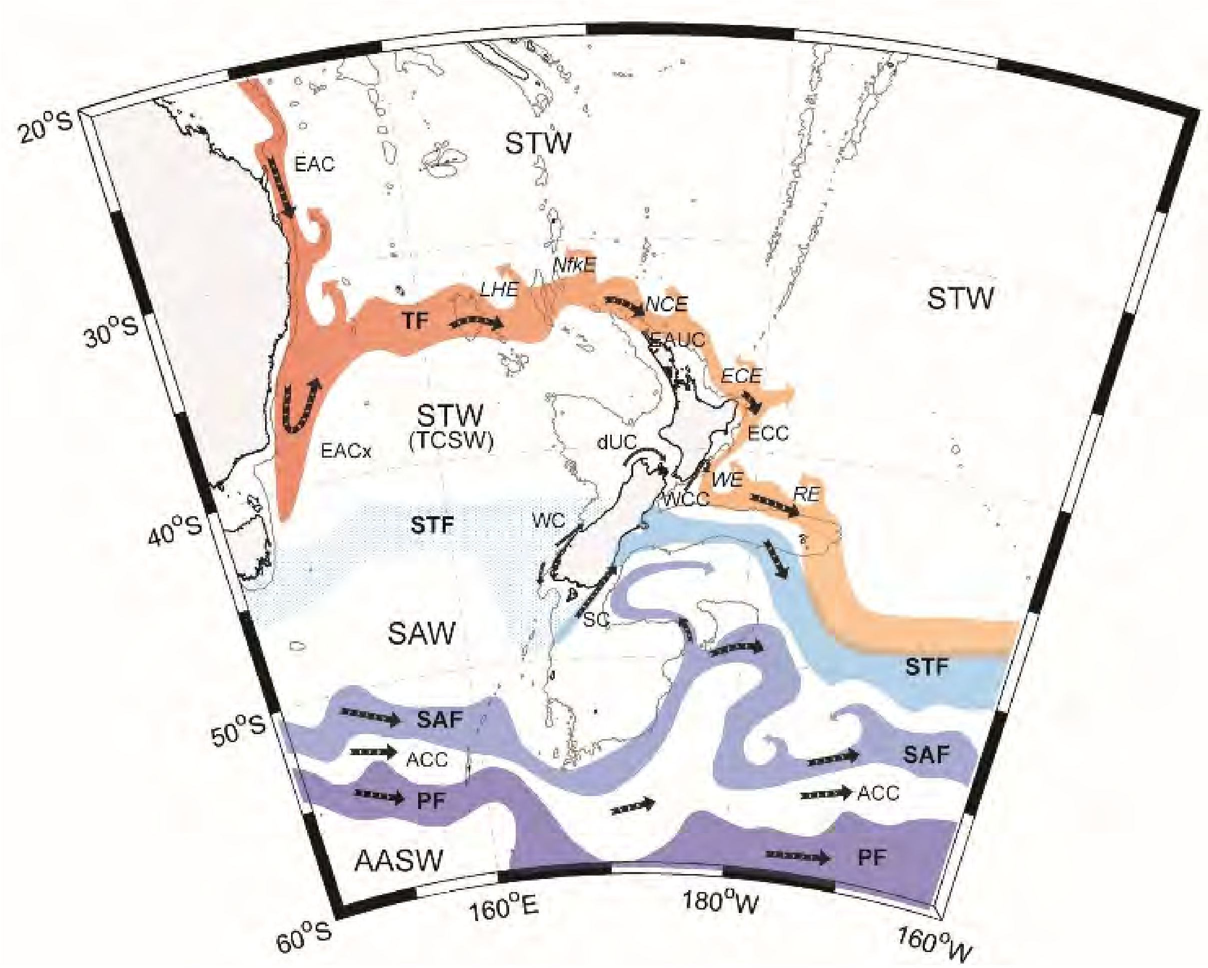


Figure 12.2: Schematic surface circulation around New Zealand based on drifter and hydrographic data. Regions of flow are shown as coloured streams. Colours reflect the temperature of the flows with red being warmest, and dark blue being coldest. The STF in the Tasman Sea is density compensated with 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).

At the southern limit of the Tasman Sea is the Subtropical Front, which passes south of Tasmania and approaches New Zealand at the latitude of Fiordland (Stanton & Ridgway 1988, Hamilton 2006). Around 165°E it diverts south across Macquarie Ridge where the front has two clear branches (Smith et al. 2013), which continue onto the Campbell Plateau where they merge as the Southland Front along the Otago Coast (Chiswell 1996, Sutton 2003).

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 is not very energetic.

This is in contrast with the east coast of both islands, and Cook Strait, which are highly energetic. Along the north-east coast of North Island there are two near permanent eddies that vary in size and strength, the North Cape and East Cape Eddies. These both sit within the EAUC, which flows down the east coast of the North Island to East Cape. From here most of the water continues south in the East Cape Current, while the remainder is 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 the Chatham Rise (Chiswell et al. 2015).

Along the south-eastern coast of the South Island the Subtropical Front flows north (while locally known as the Southland Front) to approximately the latitude of Banks Peninsula. It then turns east along the Chatham Rise before flowing south of the Chatham Islands into the Pacific Ocean.

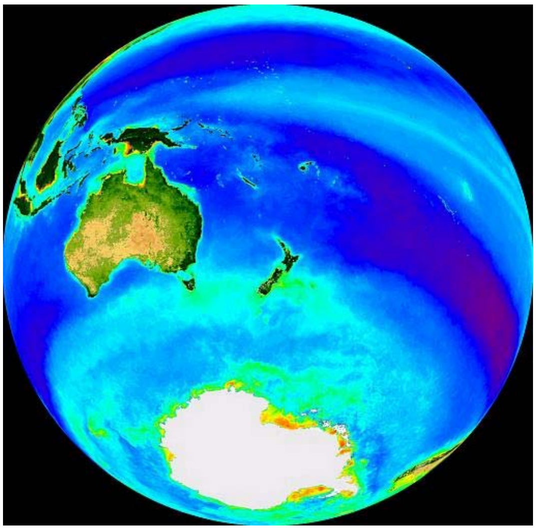
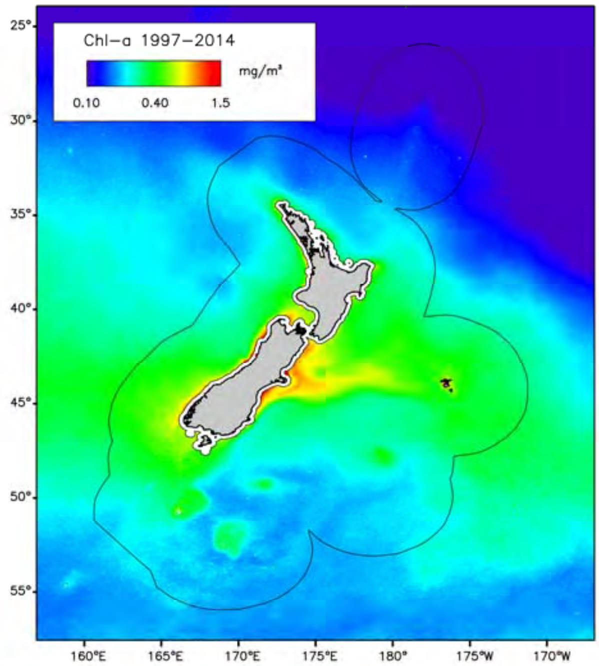


Figure 12.3: SeaWIFS image showing elevated chlorophyll a (green) near New Zealand. Image courtesy of NOAA.

Forcén-Vázquez (2015) showed that water from the Subtropical Front is found over a large fraction of the Campbell Plateau and can vary significantly from year to year. They also found water from the Southern Ocean is mixed onto the plateau, giving it a distinct oceanography from the surrounding seas. These waters are well mixed and are also 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 current flow 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). New Zealand 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).



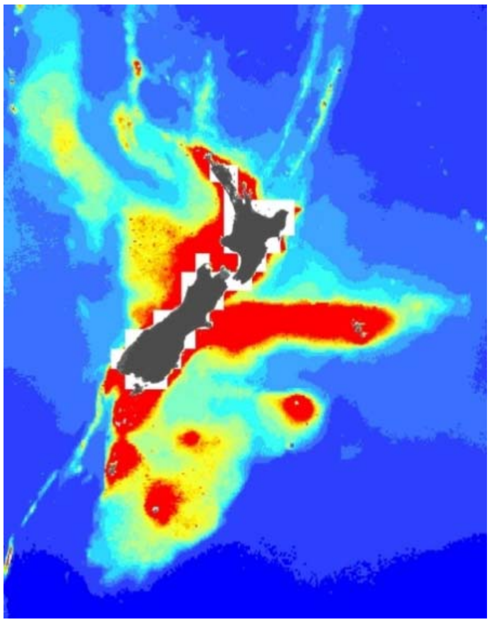


Figure 12.4: [Left panel] Mean annual concentration of chlorophyll-a (the ubiquitous phytoplankton pigment) from satellite ocean colour observations. The image is based on the merged dataset of SeaWiFS and MODIS-Aqua, which covers the period 1997–2014. The boundary of the New Zealand EEZ is also shown. Data in the territorial sea (12 n.m from the coast) is excluded because of possible contamination by suspended sediment and river run-off. [Right panel] The relative concentrations of particulate organic carbon (POC) that reach the seafloor. Red shows the highest levels, which are likely to be associated with areas of enhanced benthic productivity (based on the model of Lutz et al. 2007). Images courtesy of NIWA.

It is clear that the Chatham Rise has the highest productivity levels in the region, and this is associated with the Subtropical Front and mixing across the front. Macronutrient limited subtropical water (Bradford-Grieve et al. 1997) mix with high nutrient low-chlorophyll (NHLC) iron-limited subantarctic waters (Boyd et al. 1999) effectively removing both the iron and macronutrient restraints within the mixed waters of the Subtropical Front. The increased productivity resulting from the mixing can be seen in the high chlorophyll a detected with ocean colour remote sensing (Figure 12.3 and Figure 12.4 left panel).

Around the coast turbulence and upwelling play a key role in driving primary productivity (Figure 12.3; Pinkerton et al. 2005). However, care is needed in interpreting remotely sensed ocean colour as algorithms cannot readily distinguish between primary productivity (from phytoplankton) and sediments in freshwater runoff. Thus interpretation of the relative productivity levels inshore has to be made in conjunction with knowledge of river flow.

The high productivity in coastal waters and on ocean plateaus support a range of commercial shellfish and finfish fisheries from the shoreline to depths of about 1500 m. Seamounts, seamount chains and ridge structures in suitable depths can also provide additional localised areas of upwelling and increased productivity sometimes associated with commercial fisheries.

Patterns in surface waters of primary productivity are mirrored to an extent in the amount of 'energy' that sinks to the seafloor (Figure 12.4 right panel). This POC flux is based on a model which accounts for sinking rates of dead organisms and predation in the water column (Lutz et al. 2007). This is a potential surrogate for benthic production, and indicates where bentho-pelagic coupling may be strong. Highest levels of POC flux match with surface productivity to a large extent, with coastal waters (including around the offshore islands) and the Chatham Rise having the highest estimated production (Figure 12.4 right panel).

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 depth, all exhibit some correlation with each other over seasonal and inter-annual time scales (Hurst et al. 2012). New Zealand air temperatures have increased by about 1°C since 1900 (Figure 12.5).

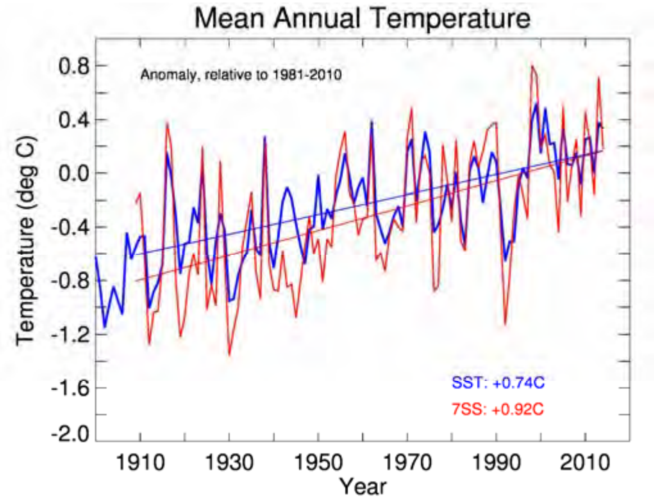


Figure 12.5: Annual time series in New Zealand. NOAA annual mean sea surface temperatures averaged over the box outlined in Figure 12.6 (blue line) and NIWA's seven-station annual mean air temperature composite series (red line), expressed as anomalies relative to the 1981–2010 climatological average. SST data from http://www.esrl.noaa.gov/psd/data/gridded/data.noaa.oisst.v2.html. Linear trends over the period 1909–2014, in °C/century, are noted under the graph. Source: updated from Mullan et al. 2010.

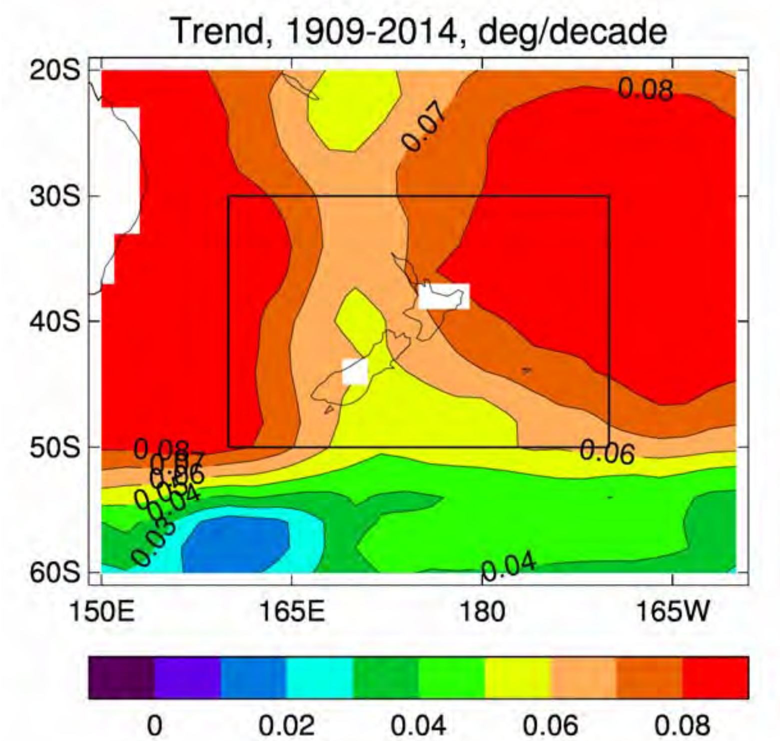


Figure 12.6: Trends in sea surface temperature, in °C/decade over the period 1909–2014, calculated from the NOAA_ERSST_v3 data-set (provided by NOAA's ESRL Physical Sciences Division, Boulder, Colorado, USA, from their web site at http://www.esrl.noaa.gov/psd). The data values are on a 2° latitude-longitude grid. Source: updated from Mullan et al. 2010.

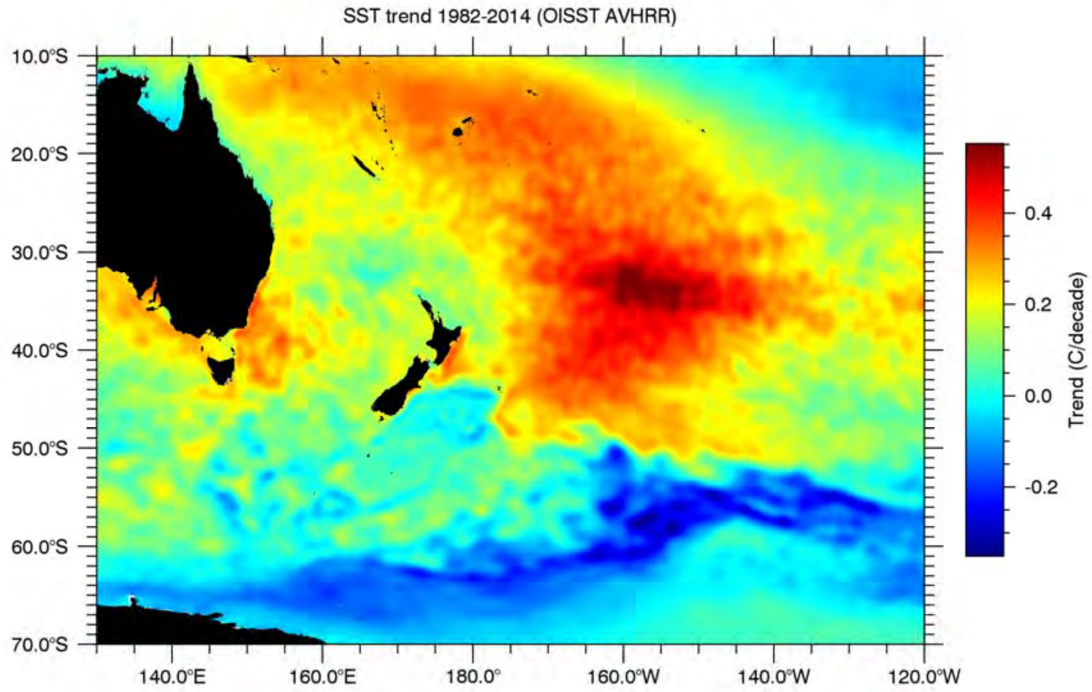


Figure 12.7: Trends in sea surface temperature, in °C/decade over the period 1982–2014. The data are from NOAA based on daily interpolated satellite measurements over a 0.25° grid. See http://www.ncdc.noaa.gov/oisst. After Reynolds et al. 2007.

Although a linear trend has been fitted to the seven-station temperatures in Figure 12.5, the variations in temperature over time are not completely uniform. For example, a markedly large warming occurred through the periods 1940–60 and 1995–2014. 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 northerly flow, and are lower in years with stronger southerly flow. One would expect this, since southerly flow transports cool air from the Southern Oceans up over New Zealand.

The unusually steep warming in the 1940–60 period is paralleled by an unusually large increase in northerly flow during this same period (Mullan et al. 2010). On a longer timeframe, there has been a trend towards less northerly flow (more southerly) 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 whole region of the Southwest Pacific within which New Zealand is situated (Mullan et al. 2010).

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 the satellite sea surface temperature trends since 1982. These are at a higher spatial resolution than Figure 12.6 providing more detail but over a shorter period. It is apparent that sea temperatures are increasing north of about 45°S; they are increasing more slowly, and actually decreasing in recent decades, off the Otago coast and south of New Zealand. This regional pattern of cooling (or only slow warming) to the south, and strong warming in the Tasman and western Pacific can be related to increasing westerly winds and their effect on ocean circulation (Mullan et al. 2010). Thompson & Solomon (2002) discuss the increase in Southern Hemisphere westerlies and the relationship to global warming; Roemmich et al. (2007) describe recent ocean circulation changes; Thompson et al.

(2009) discuss the consequent effect on sea surface temperatures in the Tasman Sea.

Coastal SST data, particularly the longer time series from Leigh and Portobello, have been used to link changes in the near shore environment with larger-scale climate signals. SST anomalies at Leigh, and at Portobello after 1968 suggest a strong relationship between SST anomalies and SOI (Figure 12.8).

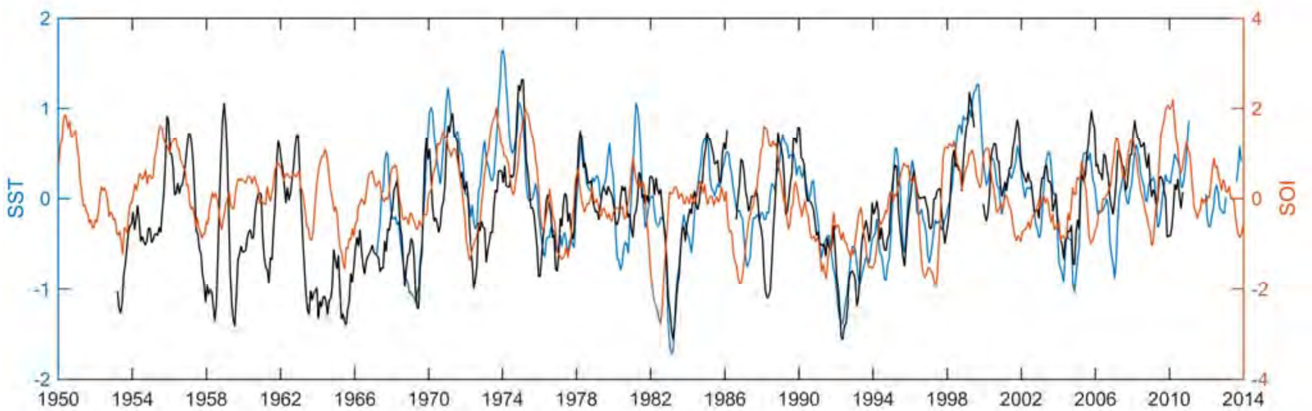


Figure 12.8: Sea surface temperature (SST) anomalies from a 40-year climatology from 1971–2010 of observations at Leigh; blue line), Portobello (black line) and the Southern Oscillation Index (SOI; red). Image updated from Hurst et al. 2012.

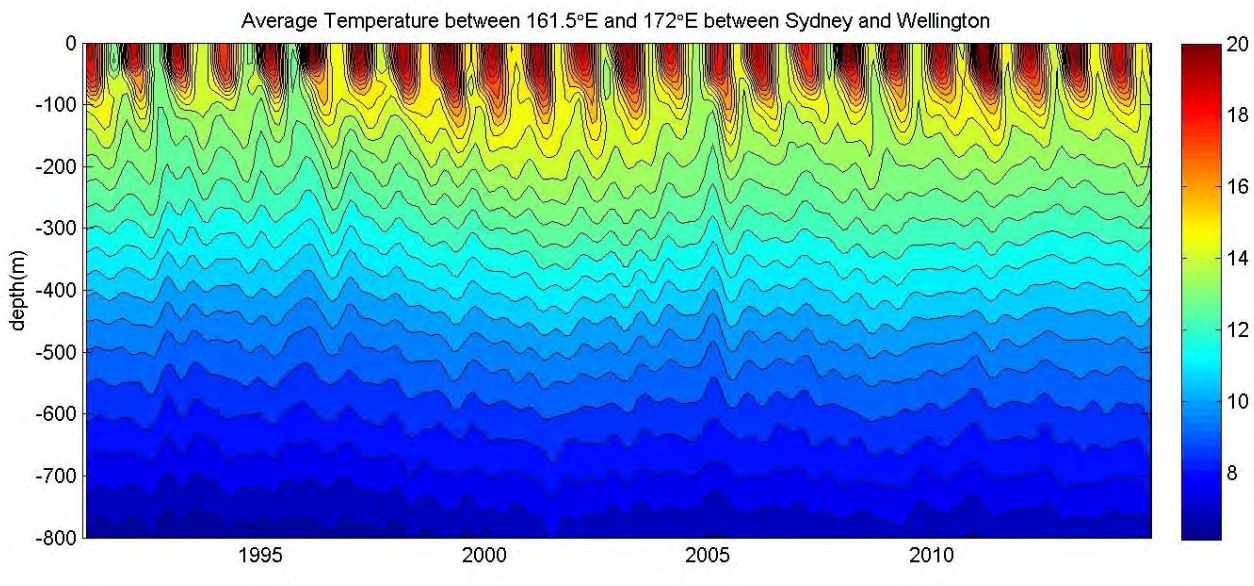


Figure 12.9: Eastern Tasman ocean temperature: Wellington to Sydney 1991–2008. Coloured scale to the right is temperature °C. Image updated from Sutton et al. 2005.

Temperature fluctuations also occur at depth in the ocean as demonstrated by changes in temperature down to 800 m in the eastern Tasman Sea between 1992 and late 2014. Variation is also seen in temperature data on the Campbell Plateau (Forcén-Vázquez 2015).

The ocean temperature between Sydney and Wellington has been sampled about four times per year since 1991. The measurements are made in collaboration with the Scripps Institution of Oceanography and CSIRO. 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 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 upper panel shows the temperature averaged along the transect between the surface and 800 m and from 1991 to the most recent sampling.

The seasonal cycle is clearly visible in the upper 100–150 m. There is a more subtle warming signal that occurred through the late 1990s, which is made apparent by 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. Finally, 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 due to an increase in high latitude westerly winds effectively speeding up the entire South Pacific gyre.

Other phenomena have led to periods of warming that are not as yet fully understood. Both stochastic environmental variability and predictable cycles of change influence the productivity and distribution of marine biota in our region.

On the Campbell Plateau, temperature data has been collected each December as part of annual trawl surveys between 2002 and 2009 and biannually since then. Forcén-Vázquez (2015) has analysed the first eight years (Figure 12.10) and found two types of variability. In the Subtropical Front near the South Island coast there was year to year variability in the temperature. On the plateau there was not only temperature variability but changes in flow patterns which were attributed to variability in the position of the Subtropical Front and influences from the Subantarctic Front.

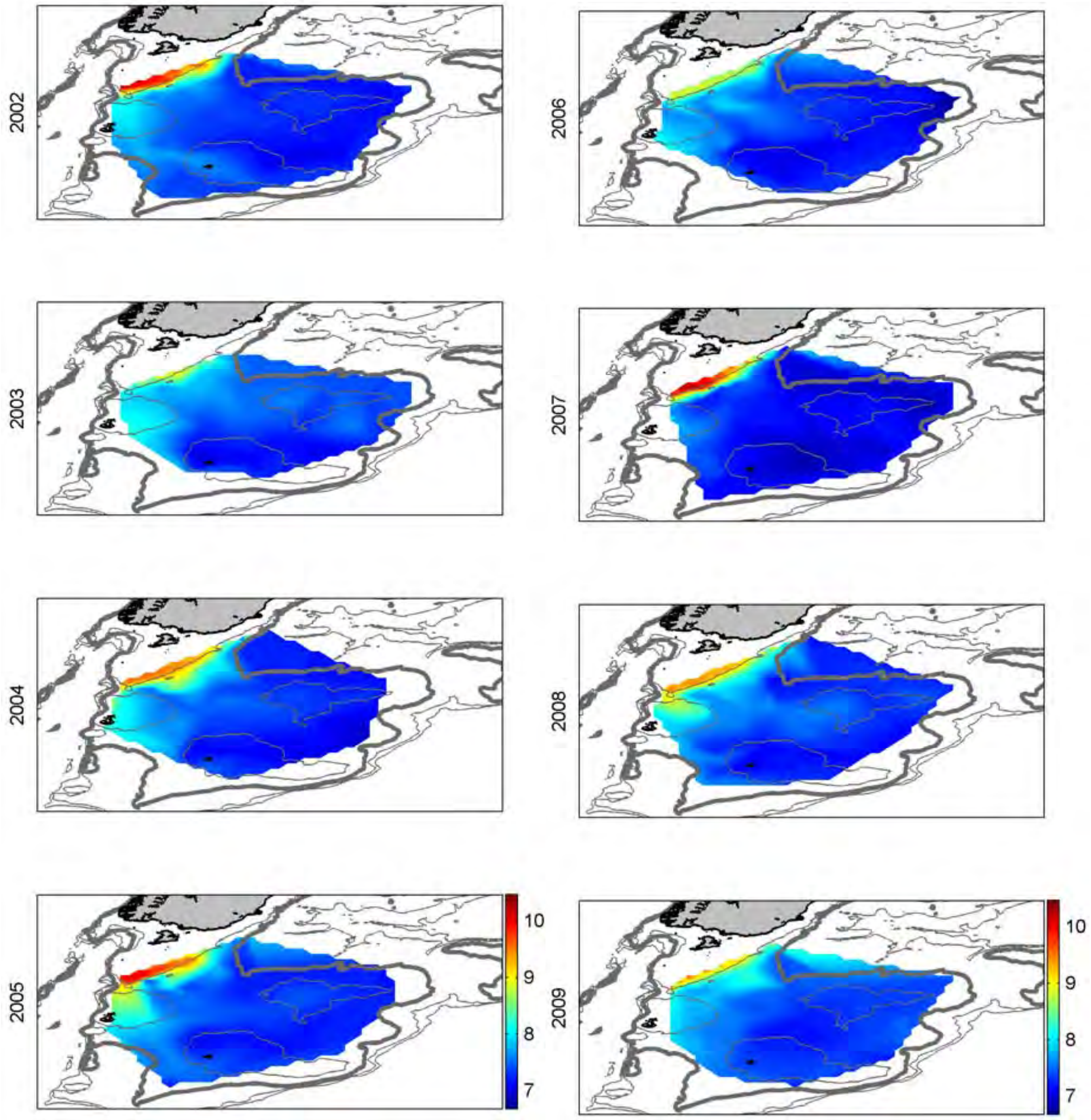


Figure 12.10: Temperature at 250 m on the Campbell Plateau interpolated from trawl mounted CTD data. Reproduced with permission from Forcén-Vázquez 2015.

12.2.2 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 (or 'regime shifts') over 15–30-year time scales. In the past 100 years, regime shifts occurred in 1925, 1947, 1977 and about 2000 (Figure 12.11). The latest shift should result in New Zealand experiencing periods of reduced westerlies, with associated warmer air and sea temperatures and reduced upwelling on western coasts (Hurst et al. 2012).

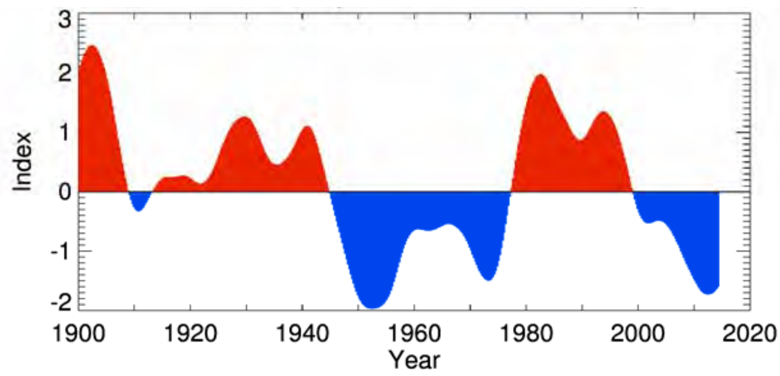


Figure 12.11: Smoothed index of the Interdecadal Pacific Oscillation (IPO) since 1900. Source: NIWA-based on data from the United Kingdom Meteorological Office, UKMO.

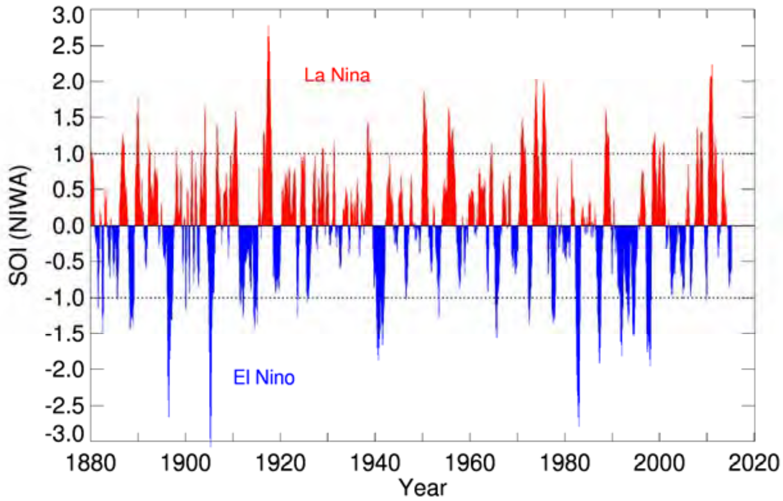


Figure 12.12: Southern Oscillation Index (SOI) five-month running mean 1880–mid 2015. Red indicates warmer temperatures, blue indicates cooler conditions for New Zealand (and the opposite in the equatorial Pacific). Image courtesy of NIWA.

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 (east Pacific) and Darwin (west Pacific). When the SOI is strongly positive (persisting above +1, Figure 12.12), a La Niña event is taking place and New Zealand tends to experience more north-easterlies, reduced westerly winds, and 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 and 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). The SOI is available monthly from 1876 onwards (Mullan 1995) (Figure 12.12).

12.2.3 WATER CHEMISTRY: OCEAN ACIDIFICATION

An 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), with global ocean uptake on the order of about 2 gigatonnes (Gt) per annum (about 30% of total global anthropogenic emissions; Rhein et al. 2013). The anthropogenic CO_2 signal is apparent to an average depth of about 1000 m.

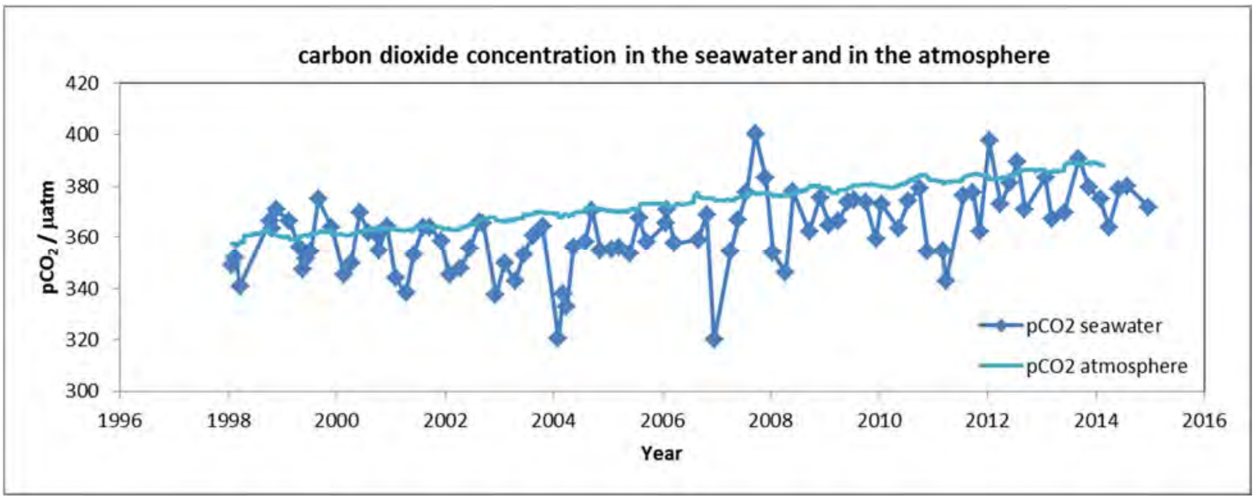


Figure 12.13: pCO_2 (partial pressure of CO_2) in subantarctic surface seawater and the overlying atmosphere from the Munida Time Series Transect, 1998–2014. Image courtesy of K. Currie, NIWA.

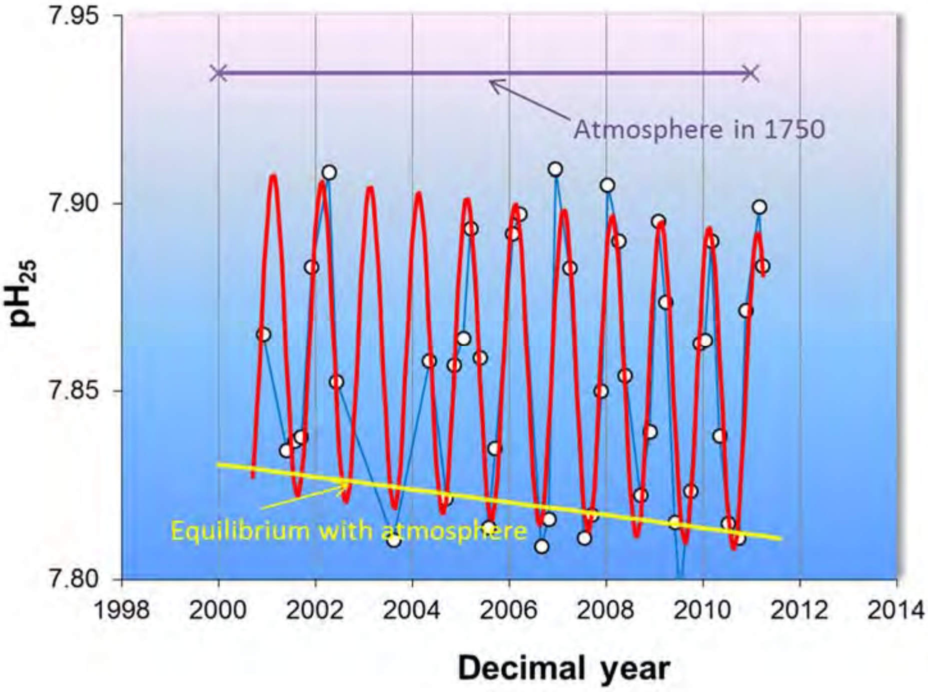


Figure 12.14: pH in subantarctic surface seawater on the Munida Time Series transect, 1998–2011. The blue points and joining lines are the actual measurements, and the red line a best fit to the points using a sine wave function (to represent seasonal change). The black line represents pH assuming equilibrium with the atmosphere concentration in the year 1750. The yellow line is the pH assuming equilibrium with actual CO_2 concentrations measured at the NIWA Baring Head Atmospheric Station. pH25 is the pH measured at 25°C. Source: A Southern Hemisphere Time Series for CO_2 Chemistry and pH K. Hunter, K.C. Currie, M.R. Reid, H. Doyle. A presentation made at the International Union of Geodesy and Geophysics (IUGG) General Assembly Meeting, Melbourne June 2011.

Carbon dioxide absorbed by seawater reacts with the water and carbonate ions to form bicarbonate ions and in the process, hydrogen ions are released. 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 0.4 pH decrease 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. In the absence of any decrease in CO_2 emissions this trend is likely to continue (Orr et al. 2005).

In New Zealand, the projected change in surface water pH between 1990 and 2070 is a decrease of 0.15–0.18 pH units (Hobday et al. 2006). The only time series of dissolved CO_2 in New Zealand waters is along a transect from the coast across subtropical and subantarctic waters off the Otago shelf – the Munida Time Series (Currie et al. 2011). Measurements have been made bimonthly by NIWA and the University of Otago since 1998. Dissolved pCO_2 shows a statistically significant increase of 1.3 μatm yr^{-1} for 1998–2012 (Bates et al. 2014) and is consistent with long-term changes observed at other global time series sites (Figure 12.13). Seasonal variability in subantarctic waters at the time series site is due to biological uptake of dissolved inorganic carbon and seasonal temperature changes (Brix et al. 2013).

The Munida Time Series pH data shows a decline in subantarctic surface waters since 1998 (Figure 12.14) consistent with that expected from equilibrium with atmospheric CO_2, as recorded at the NIWA Baring Head atmospheric station (K. Hunter, University of Otago, pers. comm.).

Surface pH in the open ocean has been determined at seven long-term time series stations, six of which are in the northern hemisphere, plus the southern hemisphere Munida Time Series site (Bates et al. 2014; Figure 12.15). All time series records show long-term trends of increasing pCO_2 (partial pressure of CO_2) and decreasing pH. Globally, open ocean seawater pH shows relatively low spatial and temporal variability, compared to coastal waters where pH may vary by up to 1 unit in response to precipitation events, diel biological activity in the seawater and sediment and other coastal processes. A New Zealand coastal ocean acidification observing network (NZOA-ON) has been set up to establish baseline conditions against which to measure future change. Biological implications of ocean acidification result from increasing hydrogen ion concentration (decreasing pH) and decreasing carbonate availability, increasing dissolved pCO_2. The concern about ocean acidification is that the resulting 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 most likely to be affected are those at the base of the food chain (bacteria, protozoa, plankton), coralline algae, rhodoliths, shallow and deepwater corals, echinoderms, molluscs, and possibly cephalopods (e.g., squids) and high-activity pelagic fish (e.g., tunas) (see Feely et al. 2004 and references therein, Orr et al. 2005, Langer et al. 2006). This is of particular concern for 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.

Ocean acidification may also impact commercial fisheries for deepwater species such as orange roughy (Clark 1999). In surface and shallow waters some phytoplankton and sea-grasses may benefit from the increase in dissolved pCO_2 due to increased photosynthesis. Early life stages are particularly vulnerable across many phyla (e.g., Byrne et al. 2013, Mu et al. 2015, Bylenga et al. 2015.).

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 behavior 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.

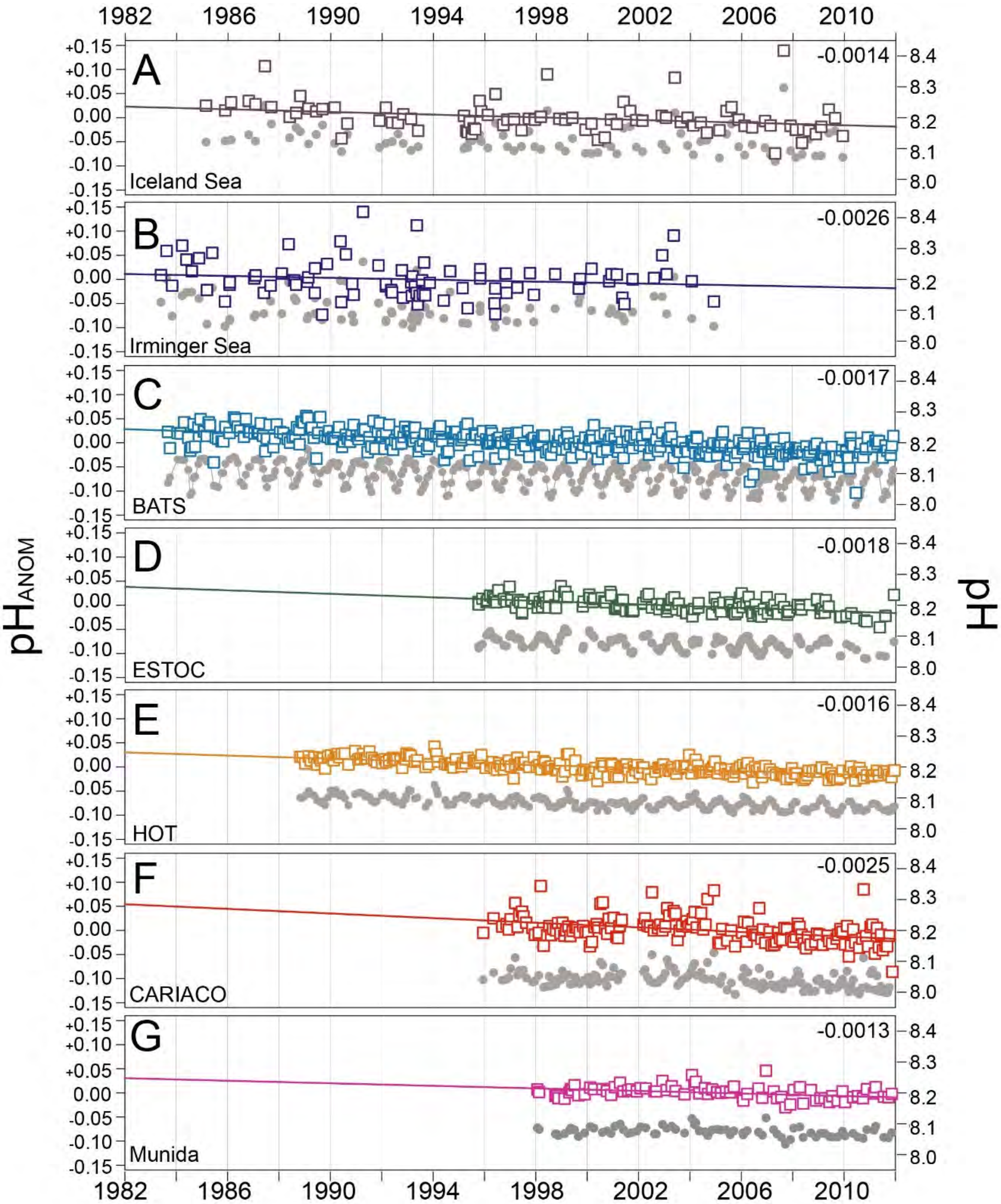


Figure 12.15: Time series of surface ocean pH (grey symbols, right hand axis) and pH anomaly (coloured symbols, left hand axis) at seven time series sites, including the Munida Time Series in New Zealand. Trends (pH change yr^{-1}) are given in the top right of each panel. (A) Iceland Sea, North Atlantic 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.

12.3 OCEAN CLIMATE TRENDS AND NEW ZEALAND FISHERIES

This section has been quoted almost directly from the summary in Hurst et al. (2012) with updates where relevant from more recent literature. 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) are:

- 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 2000. During the positive phase, from the late 1970s to 2000, 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 fishstocks 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 might be related in some way to regional warming during the period and only longer-term monitoring will 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 (see, for example, the correlation of monthly SST at Leigh and Portobello with SOI indices, Figure 12.8). 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 2000. 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 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 process 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 (Renwick et al. 1998), hake, red cod and red gurnard (Dunn et al. 2009), rock lobster (Booth et al. 2000), and southern blue whiting (Willis et al. 2007, Hanchet & Renwick 1999). The mechanisms implied by the correlations are at best poorly understood, however they motivate hypothesis testing into how wind and pressure patterns are related to fisheries.
- Temperature and sea surface height: available at least monthly over long time scales (air temperatures from 1906) or relatively short time scales (ocean temperatures to 800 m, SST and SSH variously from 1987). 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 (see Dunn et al. 2009). SST has been found to be correlated with relative fish abundance indices (derived from fisheries and/or trawl surveys) for elephantfish, southern gemfish, hoki, red cod, red gurnard, school shark, snapper, stargazer 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; most of the increase occurred 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. 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 some cooling, but not to the levels of the early 1990s.

- Ocean colour and upwelling: these will be 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 able to track changes that may have occurred since before the late 1990s warming cycle. These indices also need to be analysed with respect to SST, SSH and wind patterns, at similar locations or on similar spatial scales. The preliminary series developed exhibit some important spatial differences and trends that may warrant further investigation in relation to fish abundance indices. Of note are the increased chlorophyll indices off the west and south-west coast of the South Island in spring/summer during the last 5–6 years and the relatively low upwelling indices off the west coast South Island during winter in the late-1990s (Hurst et al. 2012).
- Currents: there are no general indices of trends or variability at present. Improvements in monitoring technology (e.g., satellite observations of SSH; CTD; ADCP; ARGO floats) have resulted in more information becoming available to enable numerical models of ocean currents to be developed. On the open ocean scale, there is considerable complexity in the New Zealand zone (e.g., frontal systems, eddy systems of the east coast). In the coastal zone, this is further complicated in coastal areas 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: The increase in atmospheric CO_2 has been paralleled by an increase in CO_2 concentrations in the upper ocean, resulting in a decrease in pH. Maintenance of the one existing New Zealand monitoring programme for pH and pCO_2, and development of new programmes to monitor the impacts of pH on key groups of organisms are critical. Potentially vulnerable groups include organisms that produce shells or body structures of calcium carbonate (corals, molluscs, plankton, coralline algae), and also non-calcifying groups including plankton, squid and high-activity pelagic fishes. Potentially positive impacts of acidification include increased phytoplankton carbon fixation and vertical export and increased productivity of subtropical waters due to enhanced nitrogen fixation by cyanobacteria. Secondary effects at the ecosystem level, such as productivity, biomass, community composition and biogeochemical feedbacks, also need to be considered.

Climate change was not specifically addressed as part of the report by Hurst et al. (2012), although indices described are an integral part of monitoring the speed 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 2000, 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 on 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 regime shift to the negative phase of the IPO in about 2000, 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 to the 1980–2000 period) and increased temperatures; this is the first regime shift to occur since most of our fisheries monitoring time series have started (the previous shift was in the late 1970s); and
- global trends of increasing air and sea temperatures and ocean acidification.

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13 TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

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 (current)	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*.
New Zealand 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. Marine biosecurity Other work on fishstocks, marine mammals, seabirds, bycatch, etc.

Note: This chapter has not been updated since AEBAR 2014.

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.

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

[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).

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 –

[2] Ministry for Primary Industries. Our Strategy. http://www.mpi.govt.nz/about-mpi/our-strategy.

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 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.

[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.

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.'

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

[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).

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 (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 13.1). 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.

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).

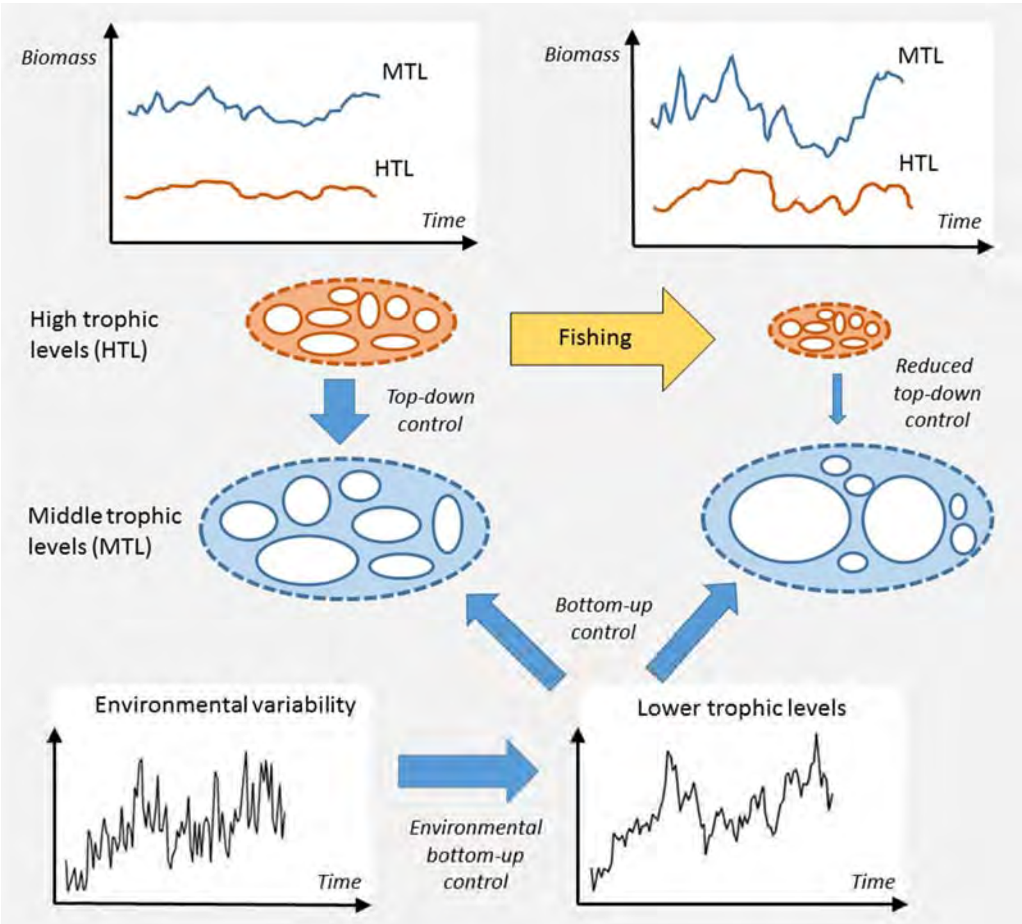

Figure 13.1: Schematic illustrating expected responses of unexploited and exploited marine ecosystems to climate forcing. Left side shows an unexploited ecosystem with multiple high trophic level (HTL) species that have relatively large abundances supported by several mid-trophic level (MTL) species, 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).

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 (

Tabl). 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 13.2: 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).

Table 13.1 [Continued]:

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.

[5] Organisms and non-living matter swimming or floating in a water body.

[6] Made locally extinct.

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 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.

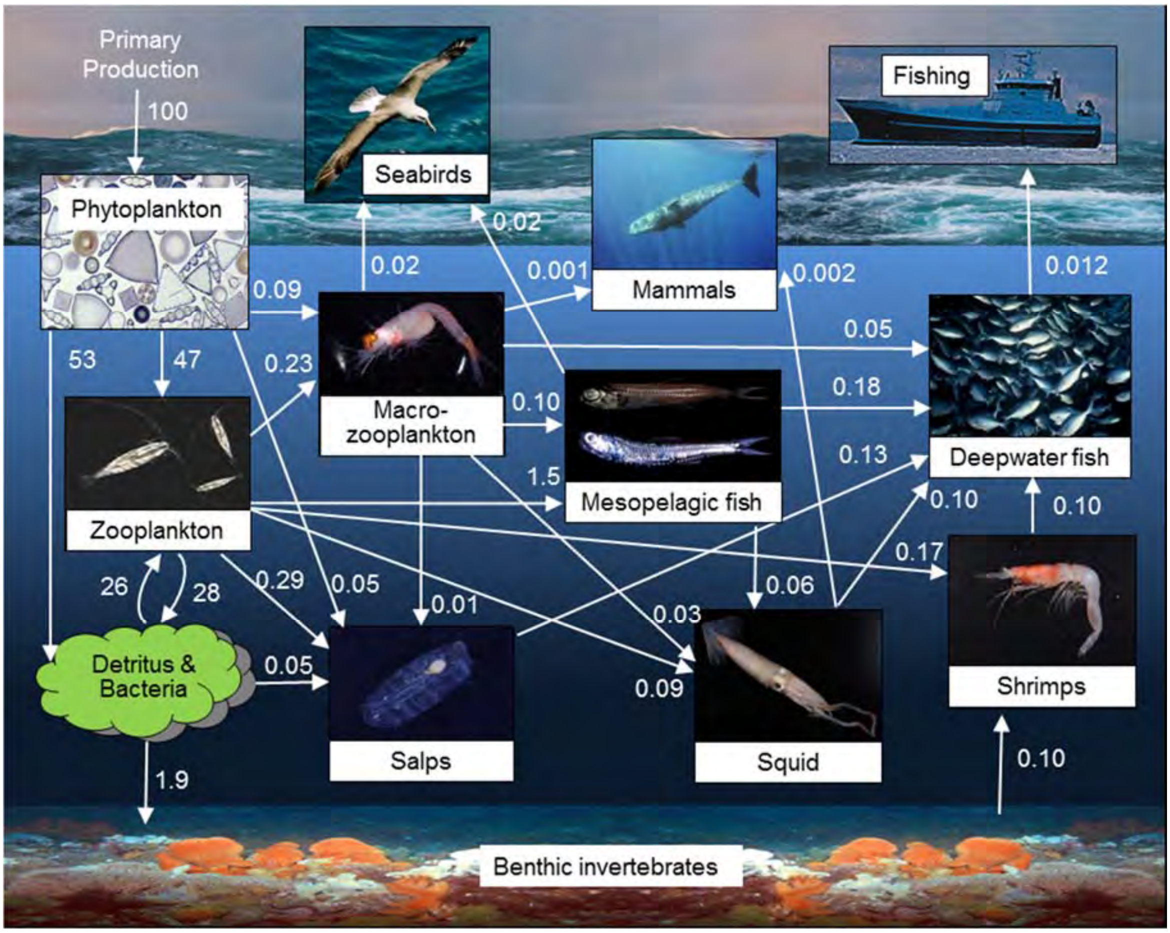

Figure 13.2: Simplified trophic model of the Chatham Rise, New Zealand (based on Pinkerton 2013). The growth of phytoplankton generates organic matter that is the fuel for the marine ecosystem. Figures show the annual flow of energy through unit area of the food web normalised to a net primary productivity (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

[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.

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 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 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

[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.

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.

[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).

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 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

[14] Species deemed more vulnerable according to the IUCN Red List (IUCN 2009); see Tuck et al. (2009).

& 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 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.

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

[15] The biomass that allows the maximum sustainable yield to be taken.

[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.

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 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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14 HABITATS OF PARTICULAR SIGNIFICANCE FOR FISHERIES MANAGEMENT (HPSFM)

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.

Note: No update has been made to this chapter since the AEBAR 2012 (apart from to reflect changes in the management implementation of HPFSM in the summary table and context sections).

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 PREDICTION OF BIOLOGICAL CHARACTERISTICS

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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15 LAND-BASED EFFECTS ON FISHERIES, AQUACULTURE AND SUPPORTING BIODIVERSITY

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, 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, consequences to seafood production of increased pollutants, 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 (current)	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.

Note: This chapter has not been updated since AEBAR 2014.

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 responsibilities regarding aquatic life (under Part 2 of the Fisheries Act). The MPI

Strategy 'Our Strategy'[1] states that New Zealand's natural resources need to be sustainable, in the primary sector.

The government's 'Fresh Start for Freshwater Programme'[2] (led by MfE and MPI) aims to create a water management 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.

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

[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.

[3] Aquaculture New Zealand. Strategy. Retrieved from http://aquaculture.org.nz/about-us/strategy.

rainfall and rates of tectonic uplift (Carter 1975), have been accelerated by human activities (Goff 1997). Sediment inputs are now high by world standards and make up about 1% of the estimated global detrital input to the oceans (Carter et al. 1996). By contrast New Zealand represents only about 0.3% of the land area that drains into the oceans (Griffiths & Glasby 1985, Milliman & Syvitski 1992).

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 bed and associated ecological communities could also affect the productivity of adjacent fishery stocks.

MPI mainly manages in the marine environment, therefore this topic area will be dealt with first. The main freshwater fisheries management MPI 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.

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

[4] NIWA. Ocean Survey 2020. Retrieved from https://www.niwa.co.nz/coasts-and-oceans/research-projects/oceans-2020.

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 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 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

[5] Department of Conservation. Māui Dolphin. Retrieved from http://www.doc.govt.nz/nature/native-animals/marine-mammals/dolphins/maui-dolphin.

[6] Ministry of Business, Innovation and Employment. Who got funded? Retrieved from http://www.msi.govt.nz/update-me/who-got-funded.

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 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

[7] CEFAS. Marine monitoring. Retrieved from http://wavenet.cefas.co.uk/Smartbuoy.

[8] Statistics supplied by New Zealand Food Safety Authority in Whangarei. Notably the fishery has not been operating since November 2012.

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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[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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16 ECOLOGICAL EFFECTS OF MARINE AQUACULTURE

Scope of chapter	The known effects of current impacts from aquaculture operations in New Zealand.
Area	All of the New Zealand EEZ and territorial sea, although presently aquaculture operations are located coastally.
Focal localities	Northland, Coromandel, Auckland, Marlborough Sounds, Tasman and Golden Bays, Canterbury, Southland.
Key issues	Uncertainty in predictions, cumulative effects, levels of nitrogen loading in coastal areas that will cause adverse effects
Emerging issues	Marine spatial planning, Integration of monitoring datasets.
MPI research (current)	ENV2012-01 *Nitrogen levels and adverse marine ecological effects* Aquaculture Planning Fund 12/03 *Marine Management Model (Waikato Regional Council)*; 12/04 *Guidance for aquaculture monitoring in the Waikato region*; 13/01 *Marlborough Sounds Hydrodynamic & Ecological Modelling*; 13/02 *Aquaculture Zoning in the Southland Region.*
NZ research (current)	C01X0904 *NIWA Sustainable Aquaculture*
Related issues	Land-based effects, marine biodiversity, habitats of particular significance for fisheries management.

Note: This chapter has not been updated since AEBAR 2013 (apart from the inclusion of information on benthic best management practices in Section 16.3.4.3).

16.1 CONTEXT

Aquaculture is the world's fastest growing primary industry and in 2011 supplied 41.2% of the supply of seafood globally, including 12.5% from marine aquaculture in the same year (FAO 2012). Fish convert a greater proportion of the food they eat into body mass than livestock and therefore the environmental demands per unit biomass or protein produced are lower (Hall et al. 2011). The production of 1 kg of finfish protein requires less than 14 kg of grain compared to 62 kg of grain for beef protein and 38 kg for pork protein. However, although farmed fish may convert food more efficiently than livestock there are important issues globally with respect to farming carnivorous fish species, which places demands on the use of capture fisheries for animal feeds.

In 2011 the Oceania region (which includes New Zealand and Australia) produced only 0.3% of the world's aquaculture production (183 516 t); globally nearly 60 million t were produced (FAO 2012). The average annual value of New Zealand aquaculture exports from 2008 to 2012 has been dominated by green-lipped mussels ($197 million), Salmon ($61 million) and Pacific oysters ($16 million) (Aquaculture New Zealand 2012). As of December 2011, aquaculture activities in New Zealand take place within approximately 19 268 ha of allocated water space (Aquaculture New Zealand 2012). This space can be categorised as below (Aquaculture New Zealand 2012):

- 7743 ha is granted to the aquaculture industry with the right to farm for a defined term, and is in known productive growing areas;
- 8960 ha is in open-ocean sites where productivity is yet to be proven;
- 1195 ha is in near shore sites yet to be developed;
- 1370 ha is undeveloped space in interim Aquaculture Management Areas (AMAs).

In New Zealand, the majority of aquaculture activities are located in the coastal marine environment, and the main current aquaculture locations are shown in Figure 16.1.

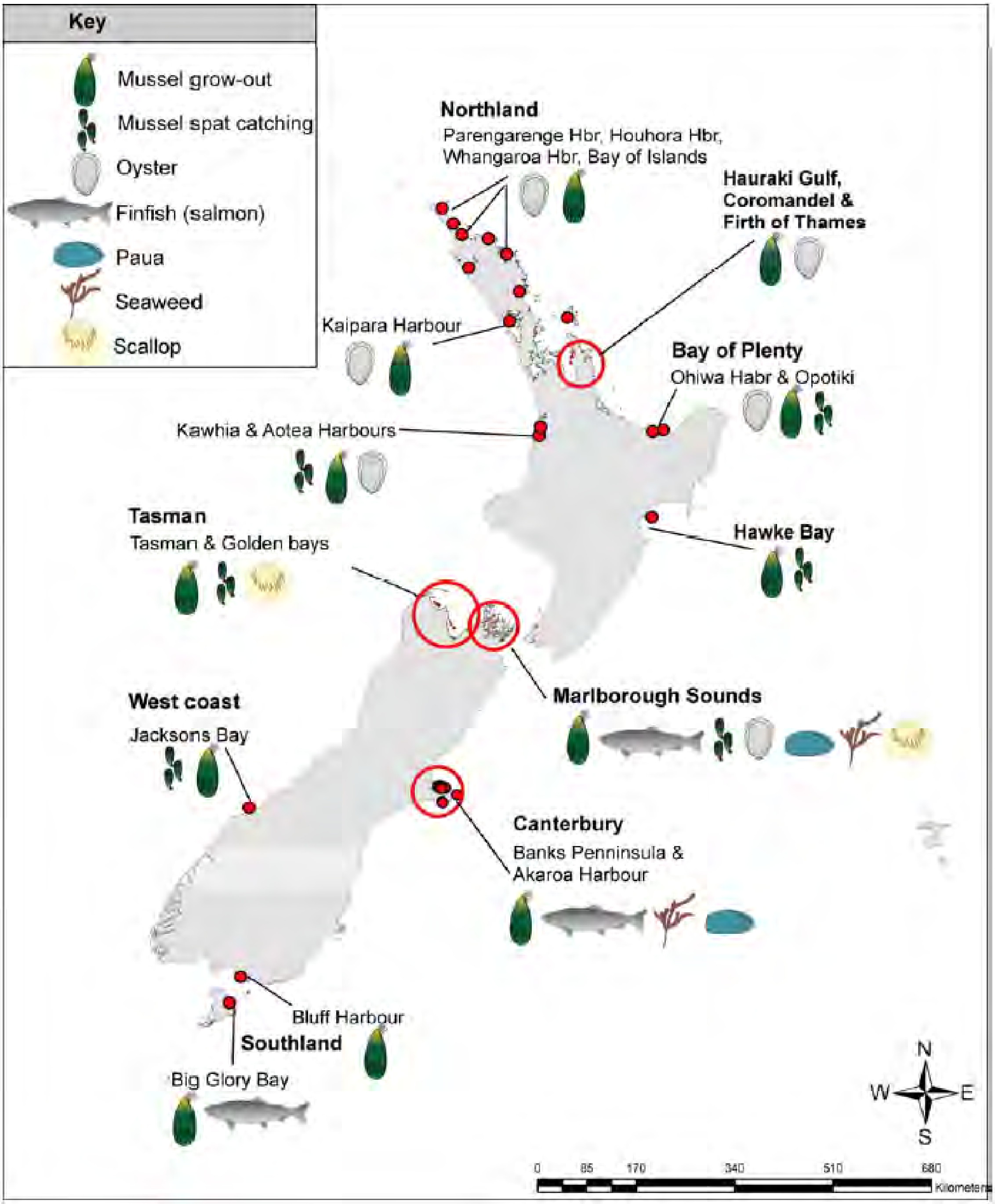

Figure 16.1: Geographic locations of main marine farming areas in New Zealand (Keeley et al. 2009).

The New Zealand aquaculture industry has a current estimated value in excess of $400 million and an objective of developing into a billion dollar industry by 2025 (Aquaculture New Zealand 2012). This ambition has been supported by the New Zealand government through the establishment of the Aquaculture Unit (now within the Ministry for Primary Industries (MPI)), the release of Government's Aquaculture Strategy and 5-Year Action Plan to support aquaculture, the 2011 aquaculture legislation reforms, and ongoing reforms of the Resource Management Act (RMA). One of the desired outcomes of these actions was to improve the consenting process to enable more space to be made available for aquaculture. To this end a number of Aquaculture Planning Fund projects have been initiated to address factors limiting aquaculture growth regionally. It is however recognised that aquaculture development, along with all other activities controlled by the RMA, needs to be ecologically sustainable.

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 therefore funded a collaborative project between NIWA and the Cawthron Institute to review the ecological effects of aquaculture (PRM2010-36). This chapter largely summarises the findings of that larger document (MPI 2013), which should be referred to for further details, references or clarification.

16.2 GLOBAL UNDERSTANDING

It is known that the environmental effects of aquaculture vary by country, region, production system and species (Hall et al. 2011). Ninety-one percent of the world's aquaculture production comes from Asia and only 0.3% from Oceania (Hall et al. 2011); therefore global reports on the environmental impacts of aquaculture tend to focus on Asia. The relevant (as judged by the authors of MPI 2013) references to New Zealand from overseas literature will hence be included in Section 16.3.

16.3 STATE OF KNOWLEDGE IN NEW ZEALAND

A 2009 survey of experts assessed the relative importance of 62 threats on 65 of New Zealand's marine habitats (MacDiarmid et al. 2012). Threat scores were categorised as extreme if the score was 3 or more, major if the score was 2–2.9, moderate if the score was 1–1.9, minor if the score was 0.5–1.0, and trivial if the score was less than 0.5. For example, the three top threats identified across all habitats were ocean acidification, increased sea temperatures from climate change and bottom trawling, which scored mean impacts across all habitats of 2.6 (major), 1.6 (moderate) and 1.5 (moderate), respectively. The study considered three threats posed by aquaculture activities: benthic accumulation of debris (shells, faeces, and food material), a decrease in the availability of primary production downstream of the marine farm (particularly mussel farms) and an increase in habitat complexity that may be detrimental to some species. The benthic accumulation of shells, food and faeces from aquaculture ranked 19th equal with a score of 0.7 (minor). The two other aquaculture threats were ranked 36th equal with a score of only 0.4 (trivial). Notably this is an average score across all habitats, however the highest scores attained for any of these aquaculture threats in particular habitats were 2.6 and 2.3 for the benthic accumulation of debris (shells, faeces, food material) in muddy sediment on sheltered coasts (2–9 m) and seagrass meadows in harbours and estuaries, respectively. The benthic accumulation of debris was the fourth most highly scoring threat in sheltered muddy coasts (2–9 m deep) and the third most highly scoring threat in seagrass meadows in harbours and estuaries.

The actual and potential effects of filter feeding and feed-added culture are shown diagrammatically in Figure 16.2 and Figure 16.3.

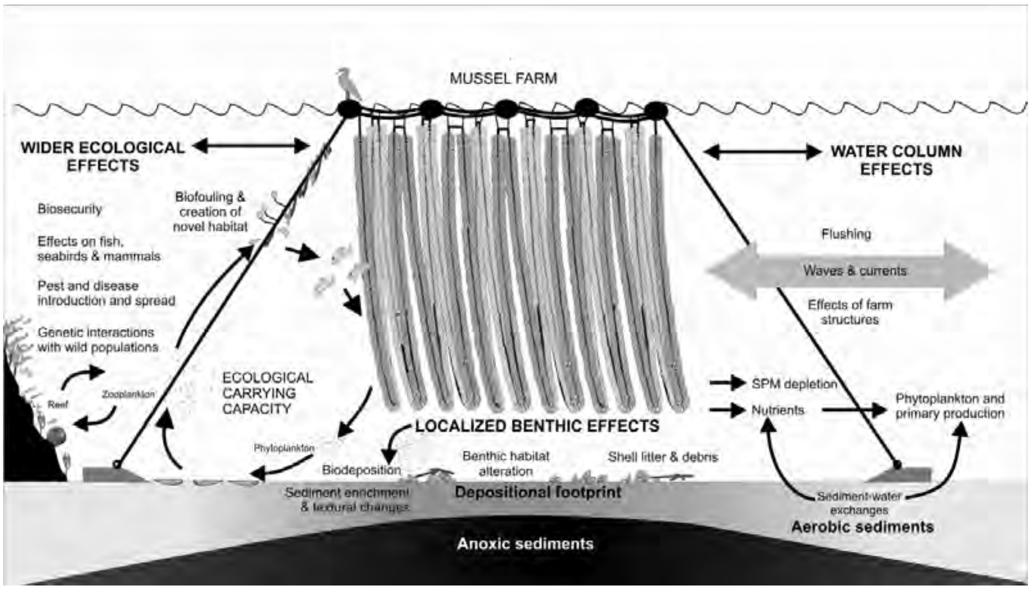

Figure 16.2: Schematic of actual and potential ecological effects from mussel farming (Keeley et al. 2009).

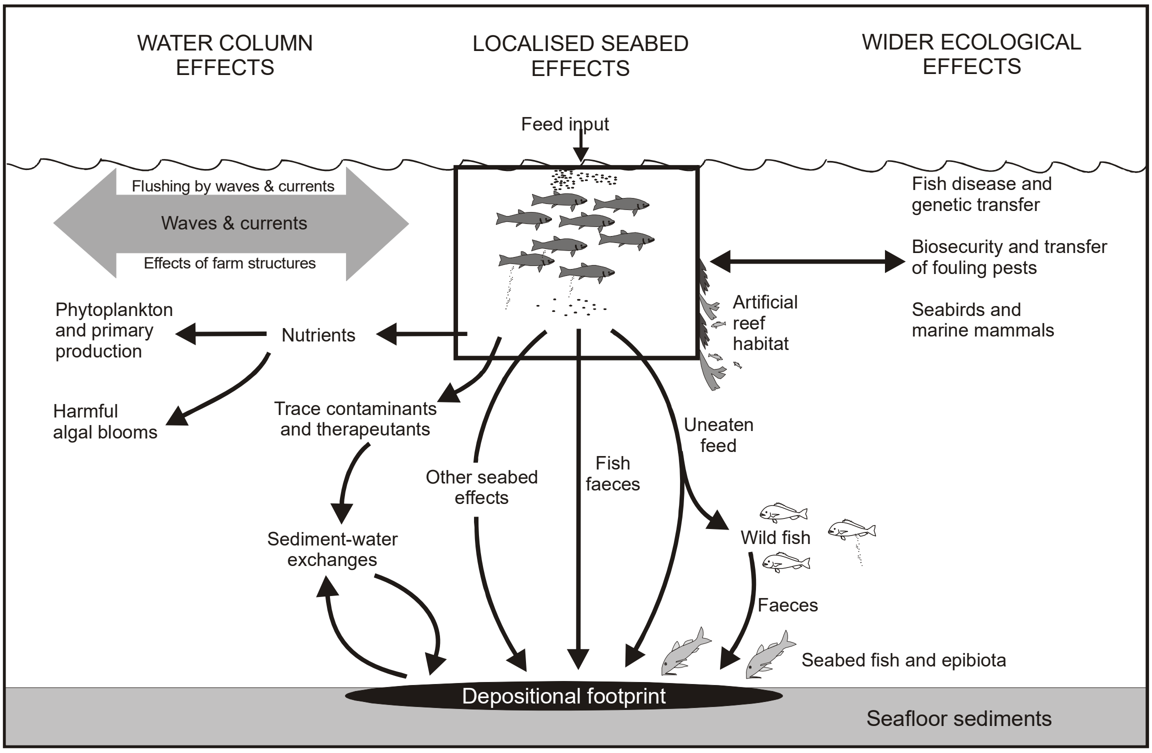

Figure 16.3: Schematic of actual and potential ecological effects from feed-added farming (Forrest et al. 2007c).

An expert panel approach was also used to trial a method for prioritising the ecological threats from aquaculture (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 threats from aquaculture. The results of this process are only indicative 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 bullet pointed below the top three, along with an explanatory sentence about what was considered under each term (in no particular order). 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 – The effect of chemicals used in aquaculture upon the environment.
- Escapee effects – the 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.

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

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 natural ongoing natural processes (see Figure 16.4 for an example of this). The interacting and cumulative effects of aquaculture will be discussed in Section 16.3.10 of this chapter.

16.3.1 BIOSECURITY THREATS

Aquaculture biosecurity has recently been covered by the reviews of Forrest et al. (2011) for finfish and Keeley et al. (2009) for other species, and then compiled and summarised in MPI (2013), this section draws heavily from those sources, and the reader is referred to them for more detail.

16.3.1.1 INTRODUCTION

The Ministry of Agriculture and Forestry (MAF) Biosecurity Strategy defines biosecurity as 'the exclusion, eradication or effective management of risks posed by pests and diseases' (Biosecurity Council 2003). 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 or economic values (e.g., the sea squirt *Styela clava*, and the red seaweed *Grataloupia turuturu*). In an aquaculture context, biosecurity also encompasses the protection of hatchery or culture operations from parasites, microscopic pathogens [2] or biotoxin-producing microalgae. These organisms may include 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). However, 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).

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

[1] Notably there was a chapter in 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.

[2] Defined here as an agent of disease, e.g., a bacterium or virus.

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 eradication,
- acceptance of some collateral environmental damage,
- follow-up monitoring to verify the completeness of the eradication.

Environmental factors including depth, wave climate, temperature regime, and currents that influence dispersal of waste, disease agents, and pests play a significant role in determining the potential biosecurity risk for a given site.

The hydrodynamics (water movement patterns that are dependent on depth, wave climate and currents) at a site play an important role on several levels. Hydrodynamics can influence the mineralisation of wastes and nutrient release through oxygen supply to the sediment and also dispersion of pathogens and pests and parasites in the water column (Zeldis et al. 2011b). 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, as 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).

Temperature and salinity can also affect the associated biosecurity risks associated with individual species by controlling their range. For example in the case of the proliferation of invasive Pacific oysters, the southern distribution is limited to Nelson/Marlborough, as 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 (Handley, unpub. 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 fouling populations is also greatest (Handley, unpub. data.).

Over the last decade 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 2002). 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 cases of data poor environments (Aguilar-Manjarrez et al. 2010, Soto et al. 2008, Silva et al. 2011).

16.3.1.2 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 level of threat 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,
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,
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. significantly reduce the flow of water (in areas of lower current velocity), carrying vital food and oxygen to cultured species.

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, unpub. 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).

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 difference species, and hence inferences regarding their direct or indirect effects (Forrest et al. 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 starvation and the recruitment failure of little penguins, which prey on pilchards (Dann et al. 2000). The potential effects of pests and pathogens are illustrated 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: likely to be new and important (***), 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).

Potentially affected uses and values	Component directly affected	Marine pests			Pathogens or parasites		
		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			?	?	?	?

16.3.1.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Biosecurity control of aquaculture activities currently occurs through: resource consent conditions, farm practices 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. Best farm practices are often described by industry codes of practice (NZMIC 2001, NZOIA 2007, NZSFA 2007). Import health standards are controlled by the Ministry for Primary Industries (MPI) 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,
- 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,
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. Both pathway management strategies have been used, for example, in relation to the New Zealand mussel industry (Forrest et al. 2011). 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 management is often guided by industry codes of practice (NZMIC 2001, NZOIA 2007, NZSFA 2007). These should include farm cleaning and surveillance (MPI 2013). Farm cleaning guidelines should deal with factors such as frequency and waste disposal. Routine surveillance, undertaken on and around marine farms is often the first point of detection of pests, pathogens and diseases.

Recent 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 fouling (Forrest et al. 2007a). To be effective, however, management requires buy-in from all marine stakeholders whose activities can spread pest organisms. Aquaculture companies can assist by:

a. identifying existing and future pests that threaten the aquaculture industry,
b. implementing surveillance of farm structures and associated vessels and infrastructure,
c. developing coordinated response plans for high-risk species before they become established,
d. preventing incursions of new pests onto aquaculture structures.

For vectors of spread such as service vessels and farm equipment, preventative management options include:

i. maintenance of effective antifouling coatings,
ii. hull inspections and hull cleaning as necessary,
iii. early eradication of pests from farm structures before they become well established.

[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.

However, once incursions have occurred, the use of eradication treatments is only advised if the risk of re-invasion can be managed. Many eradication treatments have been used in an attempt to control fouling and pests either directly (Carver et al. 2003, Coutts & Forrest 2005, Locke et al. 2009a, Morrisey et al. 2009), indirectly (Handley & Jeffs 2002, Handley 2002, Handley & Bergquist 1997) or via 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 pathological inspection of animals to ensure that infected animals are not moved into areas that do not currently have endemic infections (WWF 2010). In New Zealand, in the absence of enforced stock transfer protocols, management of 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 (*Undaria* and sea cucumbers) and their potential impacts and management measures were covered in the literature review (MPI 2013) and are summarised in Table 16.2.

Table 16.3: Matrix of biosecurity management options and their relevance to key aquaculture groups (MPI 2013). [Continued on next page]

Management measure	Description	Finfish	Shellfish	Undaria	Sea cucumbers
Harvest					
Isolate waste streams from growing areas	Prevent reintroduction of pests/pathogens to harvested sites	y	y	y	y
Fallow Sites	Reduce opportunities for reintroduction of pests/pathogens from intermediate hosts	y	y	y	y
Education					
Codes of practice	Educate and alert staff to biosecurity requirements	y	y	y	y
Public notification	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	Eradication of individual pest organisms early in the invasion process	y	y	y	y
Treatment technologies	Treatment of whole farms or bays to remove pests	y	y	y	y
Pharmaceutical treatment	Treatment of individual affected stocks to remove pathogen/parasite	y	n	n	n

Table 16.3 [Continued]:

Management measure	Description	Finfish	Shellfish	Undaria	Sea cucumbers
Import					
Import health standards	For import of seedstock	y	n	n	n
Boarder 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	Sites with appropriate environment for biological requirements of stock	y	y	y	y
Zoning	Sites location in relation to pathogen risks – other farms, processing plants, rivers, sewerage discharge	y	y	y	y
Vessel berthing	Segregate local vessels from vessels that move regionally (commercial or recreational)	y	y	y	y
Targeted surveillance	Routine monitoring for pre-determined range of species	y	y	y	y
Farm practices					
Fouling					
Management of nets, and equipment to minimise fouling	Regularly remove fouling organisms from equipment	y	y	n	n
Anti-fouling	Treat equipment with chemicals to prevent fouling	y	?	n	n
Transfer of equipment between sites/ regions	Prevent transfer of potentially contaminated equipment between sites				
Husbandry					
Appropriate stock husbandry	Minimise stress = reduce risk of disease becoming established	y	y	y	y
Management of feed so as not to attract birds/fish	Limit opportunity for transfer between sites/wild stocks through direct contact	y	n	n	n
Routine environmental monitoring linked to husbandry activities	Manage stock within environmental limits	y	y	y	y
Remove mortalities	Limit opportunity for reservoir of disease to accumulate	y	n	n	n
	Reduce attraction of predators	y	n	n	n
Use of processed feeds	Feeds heat treated to kill pests/pathogens	y	n	n	y
Surveillance	Observe and record mortality causes, unusual fouling etc.	y	y	y	y
Stock transfer		y	y	y	y
Hatchery testing for disease	Prevent diseased stock being sent to sites	y	y	y	y
Single year-class sites	Prevent disease transmission between year classes	y	n	y	y

16.3.2 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 in MPI (2013), the reader is referred to these for more detail.

16.3.2.1 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, microzooplankton 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, with sediment DO depletion resulting 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, in the case of fin-fish, or be bound in long strands composed of digested and undigested plankton, in the case of 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), the potential effects of this are usually experienced away from the farm so will be dealt with in the cumulative effects section.

16.3.2.2 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 to 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 a number of design and management factors that will greatly influence potential impacts:

- 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, 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, 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.

Undaria and sea cucumbers have less significant ecological effects on the pelagic environment since seaweeds utilise dissolved nutrients for growth (mainly dissolved inorganic nutrients (DIN)) and sea cucumbers feed on organic material on the surface of the seabed (MPI 2013). The reader is guided to the document MPI (2013) for coverage of the specific threats created via farming *Undaria* and sea cucumbers.

16.3.2.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Pelagic effects can be partially controlled through carefully selecting sites, deep sites (more than 25 m) with high currents are preferable. The farm design, orientation and stocking rates should then be appropriate to that site. Good farm management (e.g., compliance with The New Zealand Finfish Aquaculture Environmental Code of Practice (2007)[4]) 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 the cumulative effects

[4] A copy of these codes can be obtained from Aquaculture New Zealand (www.aquaculture.org.nz).

section for more discussion of these). Notably pelagic effects are reversible upon removal of the farm.

Models are an important component in determining pelagic effects at a site and a number of potential model improvements are identified in MPI (2013), including improved methods for determining ecological carrying capacity.

16.3.3 MARINE MAMMALS

The reader is referred to MPI (2013) (and references therein) for more detail.

16.3.3.1 INTRODUCTION

Several overseas studies (Würsig & Gailey 2002, Kemper et al. 2003, Wright 2008) have characterised the possible interactions between marine mammals and aquaculture, which include:

- competition for space (habitat modification or exclusion),
- potential for entanglement,
- underwater noise disturbance,
- attraction to artificial lighting,
- possible flow-on effects due to alterations in trophic pathways.

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 discussed above.

16.3.3.2 SIGNIFICANCE OF EFFECTS

Incidences of marine mammal entanglement with aquaculture operations are very few in New Zealand despite over 25 years of sea-cage salmon farming, due in part to the relatively small scale of this industry and operational procedures that minimise entanglement risk at New Zealand farms (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 while 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 prior to the farms' establishment.

These effects may need to be reconsidered in relation to any larger scale and offshore developments in New Zealand waters (MPI 2013). For instance, as multiple farms or several types of aquaculture begin to overlap or enlarge in their locations, marine mammal populations may be excluded from particular bays or regions depending on the species and its sensitivity to such activities. In the case of depleted populations (e.g., southern right whales), the issues of low population size and a fairly isolated population structure make these species more vulnerable to such impacts than other species. This large variation in the significance of aquaculture impacts (depending on the size of the affected populations) on New Zealand marine mammals makes developing and implementing one set of effective management guidelines or standards extremely difficult.

16.3.3.3 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' worth of presence monitoring was required prior to commencement of aquaculture development (Mulcahy & Peart 2012). The risks associated with physical interactions can be further minimised by adopting maintenance and operational 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 mammals. Monitoring records of the presence (and absence) of marine mammal species in the vicinity or general region of the farm site along with any detailed observations of their time spent under or around the farm structure should be compiled when possible. Future research needs to focus on those species most likely to come in contact with aquaculture in the future. In addition, ongoing research into the types of design and 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 themselves (Taylor et al. 2010).

16.3.4 BENTHIC EFFECTS

This area is covered by the review of Forrest et al. (2007c) and summarised in MPI (2013), the reader is referred there for more detail.

16.3.4.1 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);
    - in the case of intensive filter-feeder cultivation widespread biodeposition can potentially lead to a reduction in natural deposition rates;
    - smothering of benthic organisms and changes in sediment physical composition;
    - widespread biodeposition leading to mild enrichment in naturally depositional areas which has the potential for effects on reefs, inshore habitats and sensitive taxa;
    - sediment contamination (copper and zinc, covered in the additives section).
- 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;
- 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 and current speeds), although assimilative capacity may also vary seasonally in relation to factors such as water temperature. Consequently, sites located in deep water (more than 30 m) and exposed to strong water currents (more than 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).

16.3.4.2 SIGNIFICANCE OF EFFECTS

In general, benthic effects from feed-added and filter-feeder aquaculture are similar as they are caused by debris and waste falling to the seafloor generally in close proximity to the farm. However the higher volume of waste and the 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 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 this 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, the majority of the biodeposits are resuspended, exported and eventually deposited in a very diffuse form in neighbouring low flow areas (e.g., in blind bays). 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 *Undaria* and sea cucumbers are likely to be less severe on the benthos then those from feed-added or filter-feeding species (Keeley et al. 2009).

Fish farm and mussel farm studies in New Zealand and overseas indicate timescales of recovery ranging from a few months in well-flushed areas where effects are minor, to a few years in poorly flushed areas where moderate/strong enrichment has occurred (references within MPI 2013).

16.3.4.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management measures for mitigating benthic impacts for aquaculture are similar to those for mitigating pelagic impacts (Section 16.3.2.3). Site selection is important for the same reasons, to maximise the dispersive properties of the site, but should also try to avoid potentially sensitive/valuable benthic habitats (conservation areas, reefs etc.). The fine-scale positioning of the cages should optimise the dispersal of wastes and minimise impacts on potentially 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. The application of Environmental Quality Standards (EQS) can be useful. For example, best management practices (BMP) have been developed for benthic impacts from salmon farms in the Marlborough Sounds. These provide consistent and clear requirements for the management and monitoring of existing farms. Staged development and a Modelling-Ongrowing-Monitoring (MOM) approaches are also potentially beneficial (MPI 2013).

16.3.5 SEABIRD INTERACTIONS

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.5.1 INTRODUCTION

In New Zealand, the generally perceived negative effects 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 can affect things like likelihood of entanglement).

Potential negative effects may include disturbance of breeding colonies and birds feeding, blockage of the digestive tract following ingestion of foreign objects, injury or death following collision with farm structures 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). Likewise, the attraction and aggregation of small fish around marine farm structures (Grange 2002) may provide enhanced feeding opportunities for piscivorous seabirds.

16.3.5.2 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. However, there are no reports of seabird deaths as a result of entanglement in aquaculture facilities in New Zealand (Butler 2003, Lloyd 2003) as the use of top-nets over sea cages in New Zealand appears to effectively exclude seabirds (MPI 2013). 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).

16.3.5.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

At present, potential risks are identified on a case-by-case 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 (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 from different species of foraging and feeding seabirds, for example, proposed exclusion distances for king shags in the Marlborough Sounds range from 100 to 1000 m (Davidson et al. 1995, Taylor 2000), but more recently, Lalas (2001) noted that king shags resting ashore or on emergent objects only flew off when approached to within 30 m.

16.3.6 EFFECTS FROM ADDITIVES

Background data on the use and impact of chemicals locally are from research on salmon aquaculture and have been reviewed previously (Forrest et al. 2007c, 2011, Wilson et al. 2009, Burridge et al. 2010, Clement et al. 2010, MPI 2013), the reader is referred there for more detail.

16.3.6.1 INTRODUCTION

The main intentional use of additives is as antibiotics, antibacterials and other therapeutants (MPI 2013). The concern with therapeutants is their potential to affect non-target organisms (phyto- and zooplankton, sediment bacteria) and the rise of resistant bacteria and/or parasites (GESAMP 1997, Forrest et al. 2007c, 2011). The main unintentional additions are from zinc in fish feed and copper when used as an antifouling agent on structures (MPI 2013). The main concern with metals is their toxicity to animals (Forrest et al. 2007c, 2010, Clement et al. 2010).

16.3.6.2 SIGNIFICANCE OF EFFECTS

Currently, there is minimal use of chemicals such as antibiotics, antibacterials and other therapeutants intentionally added to the marine environment by the New Zealand aquaculture industry; however, culture of native species may lead to the emergence of diseases that may require new treatments.

Recent 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 between 2005 and 2010; Hopkins et al. 2006). 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 as they may not be bioavailable (Forrest et al. 2007c).

16.3.6.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

All species cultured for human consumption from aquaculture have to meet strict food safety standards, which 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.

No chemical/additives are known to be used in the farming of bivalves and lower trophic level species. If these are used in the future ‘best management practice’, should minimise food wastage and the 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 and alternative therapies, in particular vaccines, which are not presently licensed for use, nor used, in New Zealand.

The potential for environmental issues from therapeutant use in the future will need to be assessed on a case-by-case basis. Use of therapeutants in New Zealand is low, but their persistence in the environment, the induction of resistance of targeted organisms and the effects on non-target organisms are the main knowledge gaps. 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.

16.3.7 ESCAPEE EFFECTS

The subject of escapee effects from aquaculture is well covered for finfish by the reviews of Forrest et al. (2007c) for New Zealand and Jensen et al. (2010) for Norway, and for shellfish by Keeley et al. (2009) and summarised in MPI (2013). The reader is referred to these sources for more detail.

16.3.7.1 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 (hāpuku, kingfish) or introduced (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,
- 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 in terms of:

- 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,
- 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.

16.3.7.2 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 (in terms of salmon; Deans et al. 2004) and the broad home range (in terms of 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.

16.3.7.3 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.

16.3.8 EFFECTS ON WILD FISH

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.8.1 INTRODUCTION

A potential immediate effect on wild fish populations from the development of a finfish farm is the degradation or loss of habitat beneath or within close proximity to new farm structures (e.g., spatial overlap with species' critical spawning grounds and/or migration routes). By adding three-dimensional structures to the marine environment, finfish farms 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). 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 the following:

- enhanced predation on wild fish by higher trophic level predators (e.g., seals) 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 (N. Keeley, pers. comm.), which could affect wild fish populations differently than in the absence of the structures,
- 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).

16.3.8.2 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 of king salmon already in New Zealand. Modelling of larval egg depletion (Broekhuizen et al. 2002) and other work suggest that while the feeding of fish in farms 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 food added as an attractant), but shell-drop is likely to create a (lesser) attraction. The extent of impacts from the farming of *Undaria* and sea cucumbers is likely to have a lesser impact than feed-added or filter-feeding aquaculture, as they neither require feed nor exhibit shell drop (MPI 2013).

16.3.8.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management options identified in MPI (2013) for minimising 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).

16.3.9 HYDRODYNAMIC EFFECTS

The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.9.1 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,
- 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.

16.3.9.2 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 fish cages, and this will result in a shadow of reduced wave activity behind the farmed areas (Chan & Lee 2001, Lader et al. 2007).

While 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.

16.3.9.3 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.

16.3.10 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 wider ecosystem effects of aquaculture in relation to the water column is provided in Section 16.3.2: Pelagic effects. The reader is referred to MPI 2013 (and references therein) for more detail.

16.3.10.1 INTRODUCTION

The previous sections (16.3.1–16.3.9) 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 onto 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 16.3.4). 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.
- In the case of farm structures, aquaculture involving numerous farms situated along the coast could also have cumulative effects on nearshore currents and waves, which in turn could affect important processes (e.g., larval transport, nutrient exchange) along the shoreline (see Section 16.3.9). 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 16.3.1 provides comprehensive information on methods for minimising biosecurity risk that are applicable to wider, regional scales).

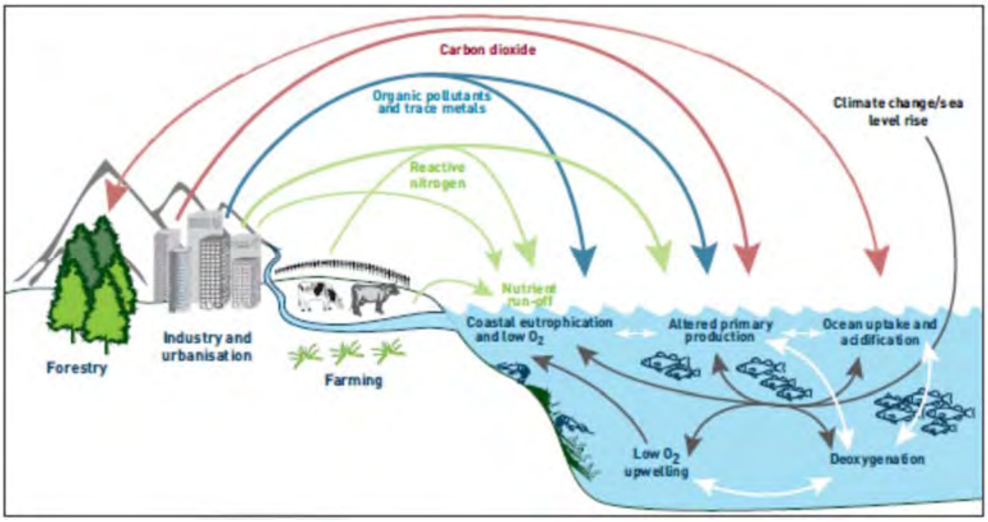

Figure 16.4: Conceptual diagram of anthropogenic influence in marine ecosystems.

Limited resources and uncertainty in understanding all of the potentially complex interactions between aquaculture, other stressors and the environment necessitates the need 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.

16.3.10.2 SIGNIFICANCE OF EFFECTS

The 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 (NH4). 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 limiting phytoplankton growth under certain conditions e.g., when concentrations are generally low and light is plentiful (MacKenzie 2004, Howarth & Marino 2006). Complicating matters is the fact that 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 Greenshell 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.

16.3.10.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

The management of cumulative effects in the marine environment can be addressed 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 (SoE) monitoring programmes. Standardised monitoring requirements for aquaculture is an important step in ensuring the usefulness of consent monitoring datasets within broader-scale assessments. The requirements for assessing and managing 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).

Two ongoing projects will help address monitoring requirements for aquaculture. An ongoing MPI Biodiversity project 'Marine Environmental Monitoring Programme' (ZBD2010-42) is seeking to address the following two objectives:

1. prepare an online inventory of repeated biological and abiotic marine observations/datasets in New Zealand,

2. review, evaluate fitness for purpose, and identify gaps in the utility and interoperability of these datasets for inclusion in a Marine Environmental Monitoring Programme (MEMP) from both science and policy perspectives.

Therefore any attempts to standardise monitoring datasets for aquaculture should try to learn from the experience or recommendations of this project. In addition the Aquaculture Planning Fund project 12/04 'Guidance for aquaculture monitoring in the Waikato Region' will develop an environmental monitoring framework to manage environmental change from aquaculture growth that will incorporate SOE monitoring, consent monitoring and predictive monitoring and have application to other regions.

Spatial modelling tools offer a way of estimating 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 (particularly within the Sustainable Aquaculture MBIE funded programme CO10X0904).

Some generalisations have been proposed in terms of 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 in order to minimise cumulative effects on the wider ecosystem. Whilst recently 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. An ongoing MPI project 'Nitrogen levels and adverse marine ecological effects' (ENV2012-01) is seeking to determine to what extent knowledge from overseas about the adverse effects of nitrogen on the marine environment can be applied here.

In the case 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). Managing cumulative effects to achieve sustainability ultimately requires regional approaches to managing 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. Using a precautionary approach, development should be conducted in a staged manner based on conservative limits of expansion. Important tools and components of a precautionary approach include:

1. The use of models and existing data to gauge limits to development[5] 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 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 for validating and improving accuracy of predictive models and understanding the role of feed-added aquaculture in driving cumulative effects.

In New Zealand the Limits of Acceptable Change (LAC) adaptive framework has been applied in the 3000 ha Wilson Bay Aquaculture Management Area (AMA), in the eastern Firth of Thames.[6] This involved stakeholders agreeing both to levels of acceptable change in indicators, and to management responses to apply if monitoring showed that these changes have been exceeded. An overseas example of the precautionary approach is the M-O-M system (Modelling–Ongrowing fish farms–Monitoring), which has been undertaken in Norway to provide information for adaptive management of salmon farming (Ervik et al. 1997, Hansen et al. 2001).

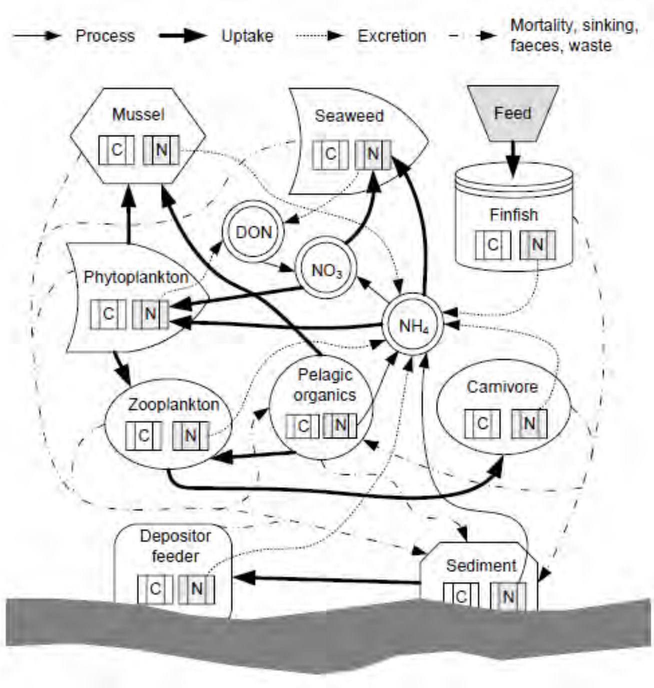

Figure 16.5: Conceptual diagram of IMTA model in terms of carbon (C) and nitrogen (N) biomass (from Ren, pers. comm.).

A precautionary approach necessitates establishment of conservative thresholds or limits to minimise risks and the extent of cumulative effects. Minimising risk of eutrophication by setting a limit (or cap) on nutrient loads in a coastal receiving environment would be similar to the approach taken in restoring the Rotorua Lakes. 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 or CNLRs (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 and Tasman Bays (Zeldis 2008b, Zeldis et al. 2011a, 2011b) and the Firth of Thames (Zeldis 2008a, Zeldis et al. 2010).

Internationally, there is a very limited understanding of the cumulative effects of multiple stressors on marine ecosystems in the long-term. A critical requirement for

[5] In some cases, areas may not be suitable for any development of aquaculture.
[6] NIWA (2006) Limits of acceptable change: a framework for managing marine farming. Retrieved from http://www.niwa.co.nz/publications/wa/vol14-no2-june-2006/limits-of-acceptable-change-a-framework-for-managing-marine-farming.

understanding these effects is having good information on existing environmental conditions, and continued monitoring to provide long time series datasets from which to validate models and quantify and forecast changes occurring in the wider environment.

Modelling has an important role to play in understanding, predicting and managing cumulative effects and New Zealand has access to extensive modelling capability; yet in most cases 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, while more advanced biogeochemical models require validated estimates of inputs (e.g., surface water, groundwater, marine) and losses (denitrification, burial rates) of nutrients specific to New Zealand's coastal waters.

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17 ANTARCTIC SCIENCE

Scope of chapter	This chapter outlines the ecosystem structure in the Ross Sea sector of the Southern Ocean (Ross Sea and Amundsen Sea), toothfish fishery in this region, nature of ecosystem-fishing interactions, the management approach, trends in key indicators of fishing effects, 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 Ross Sea shelf and slope, Amundsen Sea seamounts and shelf, 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, (5) benthic habitat.
Emerging issues	Role of Marine Protected Areas in the Ross Sea region.
MPI research (current)	ANT2017.
NZ government research (current)	MBIE: C01X1226 *Ross Sea Climate and Ecosystem*. NIWA Core Funding, Coasts & Oceans Programme 4: 'Structure and function of marine ecosystems'.
Related chapters/issues	Benthic (seabed) impacts; Trophic and ecosystem-level effects; Biodiversity. MPI 2017 Fisheries Plenary Report May 2017 Chapter 99: Toothfish.

This chapter has been updated for the 2017 AEBAR.

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 2017, Mormede 2017) and a summary is available as part of the May Plenary Report (MPI 2017). Research towards a stock assessment for Antarctic toothfish in the Amundsen Sea region is work in progress (Large et al. 2016).

Section 17 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 MPA.

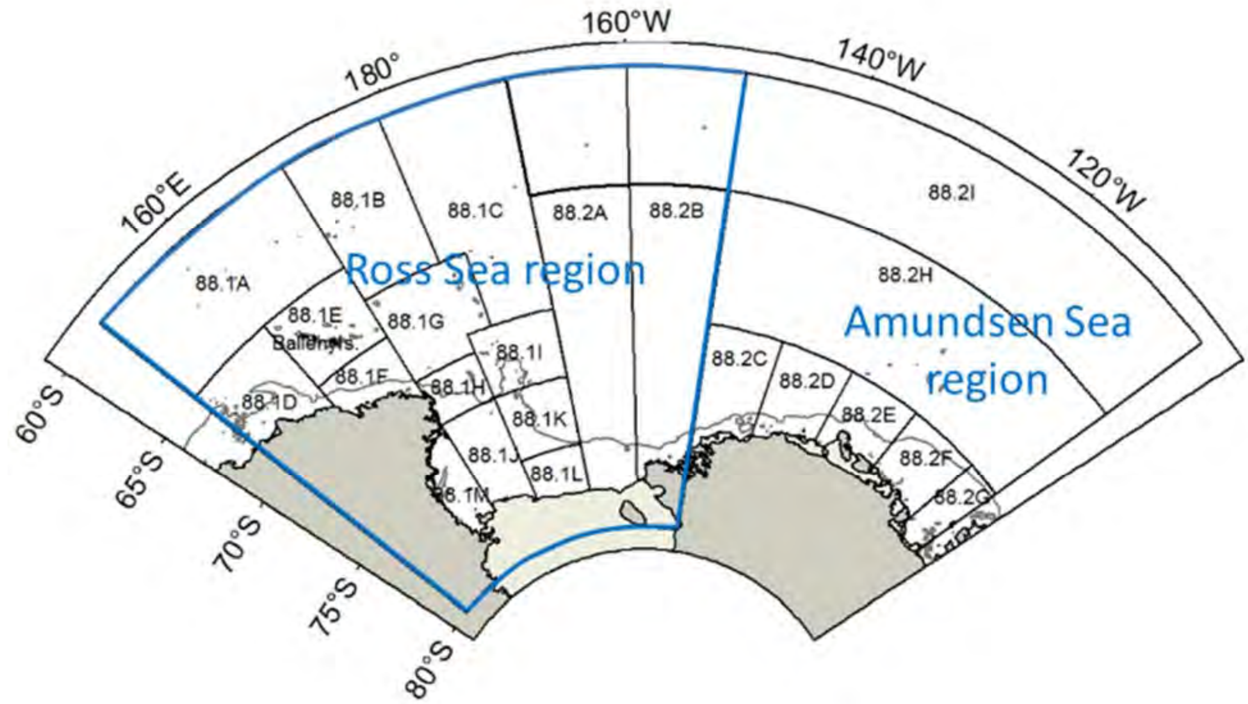


Figure 17.1: The Ross Sea and Amundsen Sea regions, which span CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) Subareas 88.1 and 88.2. CCAMLR small scale statistical areas (SSRUs) and the depth contour at 1000 m are shown. The Ross Sea region is made up of 88.1 and 88.2A, 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*).

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.

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 all seven species of whales found in the region were extensively exploited. 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

[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.

(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 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.

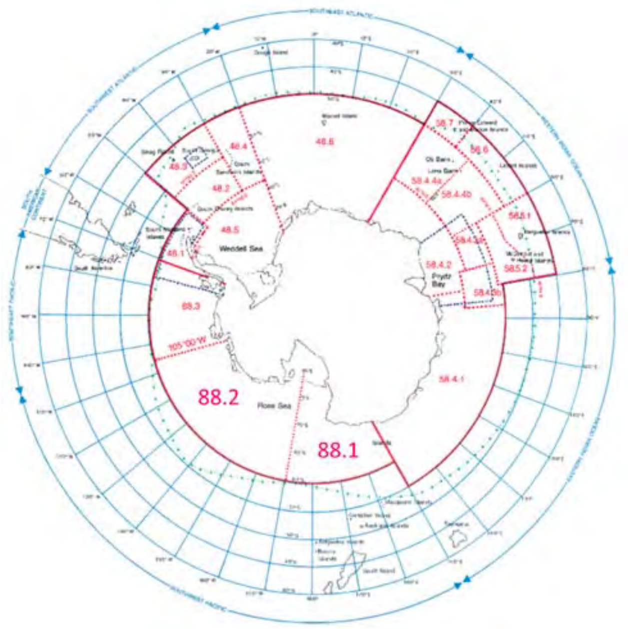


Figure 17.2: Boundary (solid pink line) of area managed according to the Convention for the Conservation of Antarctic Marine Living Resources. Dashed pink lines delineate CCAMLR statistical subareas.

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 (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 24 States[3] and the European Union (acting as a single member). A further 11 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 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).

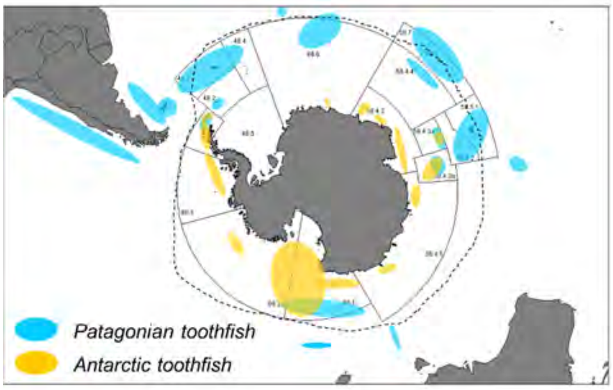


Figure 17.3: Distributions of Antarctic and Patagonian toothfish in the Southern Ocean. Approximate location of Antarctic Convergence shown by the dotted line.[7]

For Antarctic toothfish in the Ross Sea region, spawning dynamics and early life history is described in Hanchet et al. (2008).

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

[3] Argentina, Australia, Belgium, Brazil, Chile, China, European Union, France, Germany, India, Italy, Japan, Republic of Korea, Namibia, 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, Netherlands, Islamic Republic of Pakistan, Republic of Panama, Peru, Vanuatu.

[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.

and larval dispersal simulations. The study concluded that 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. 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).

More information on the life history and stock structure can be found in the Fisheries Assessment Plenary (Ministry for Primary Industries 2017).

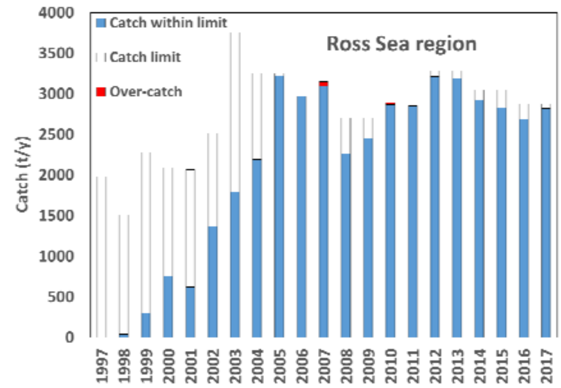
The stock structure of Patagonian toothfish in the Ross Sea and Amundsen Sea regions is less well known. Patagonian toothfish in the Ross Sea region are believed to come from a stock which is widely distributed beyond Macquarie and Campbell Plateau and into the high seas.

17.1.5 ROSS SEA REGION TOOTHFISH FISHERY

A characterisation of the fishery in the Ross Sea region is given in Parker & Mormede (2017). 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, while catches of Patagonian toothfish taken mainly from the north-west of the Ross Sea region have averaged only about 6 t/y since 2005 compared to 2860 t/y of Antarctic toothfish.

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 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.

a



b

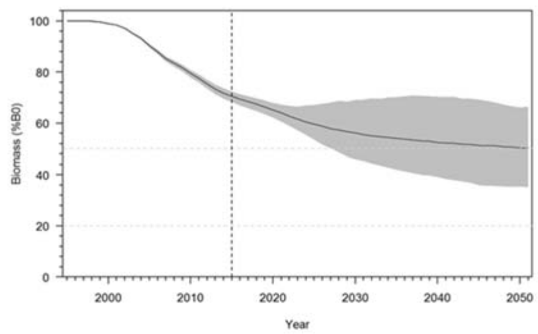


Figure 17.4: [a] Catch and catch-limit, [b] assessed spawning stock biomass (with 5th–95th percentiles in grey) for Antarctic toothfish in the Ross Sea region (CCAMLR subareas 88.1 and 88.2A, B). Antarctic fishing years are labelled as the later year of the season (e.g., the 1997–98 fishing season is labelled ‘1998’). [CCAMLR 2016a; Mormede et al. 2015, Parker and Mormede 2017].

The average Illegal, Unreported and Unregulated (IUU) catch of toothfish in Subarea 88.1 was estimated to be 50 t between 2005 and 2010. The IUU catch has not been estimated since 2010; there is no confirmed nor significant IUU presence or activity reported in the Ross Sea region (CCAMLR 2016a).

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., Hanchet et al. 2015b).

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 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.

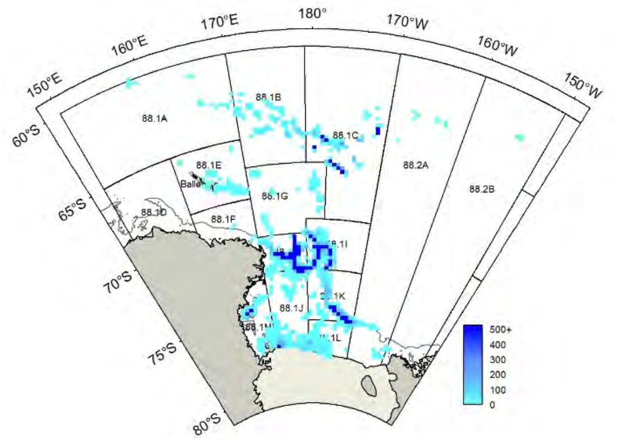


Figure 17.5: Spatial distribution of total toothfish catch in the Ross Sea region (t) from 1997 to 2015.

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 (Parker & Mormede 2017).

The Ross Sea region toothfish fishery was first certified by the Marine Stewardship Council (MSC) in November 2010 and was recertified in 2015. The MSC Fisheries Standard is designed to assess if 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 in Large et al. (2016). The toothfish fishery in the Amundsen Sea region has been operating since 2003 (Figure 17.6), with an annual catch of 350–624 t since 2006 (CCAMLR 2016b, Large et al. 2016).

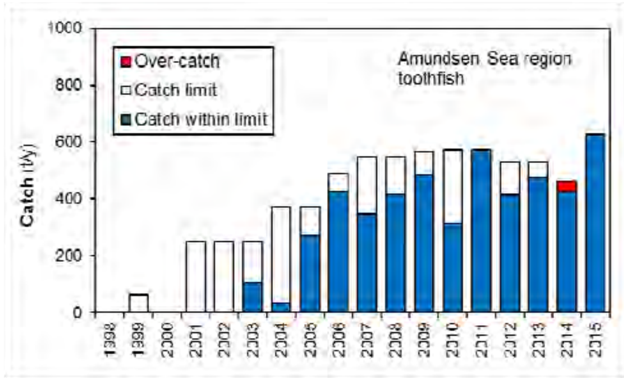


Figure 17.6: Catch and catch-limit for Antarctic toothfish in the Amundsen Sea region (CCAMLR subareas 88.2 C-I; CCAMLR 2016b).

Up to 2014 the main fishery in this area has operated in the northern SSRU (Large et al. 2016). In contrast, up to and including 2014, the catch and effort in the southern SSRUs have been low (Figure 17.7). Parker (2014) showed high local exploitation rates and indications of localised depletion on some individual seamounts in the north.

Only five Patagonian toothfish have been caught in the Amundsen Sea region since 2004.

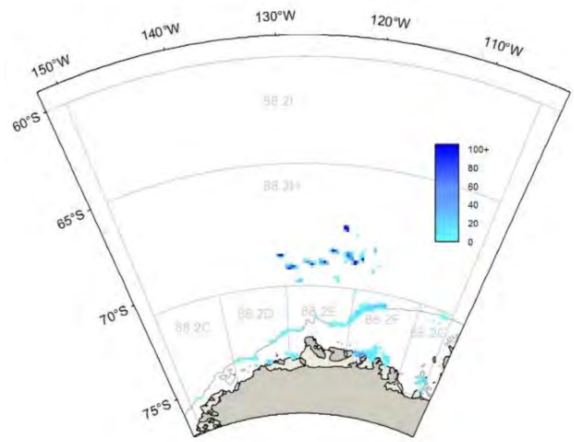


Figure 17.7: Spatial distribution of total toothfish catch in the Amundsen Sea region (t) from 1997 to 2015.

More data is required before a robust stock assessment for the Amundsen Sea region can be developed and this work is active at the time of writing (Mormede et al. 2016).

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 through 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 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 sea birds 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 immediate below the ice (Garrison 1991, Brierley & Thomas 2002, Arrigo & Thomas 2004). Present estimates suggest that the contribution of epontic[8] 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. *P. antarctica* are found in the diet of all large animals (seabirds, seals, toothed and baleen whales, toothfish, many other species of fish, 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

[8] Epontic: referring to organisms closely associated with sea ice.

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 (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 as 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 17.8) 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.

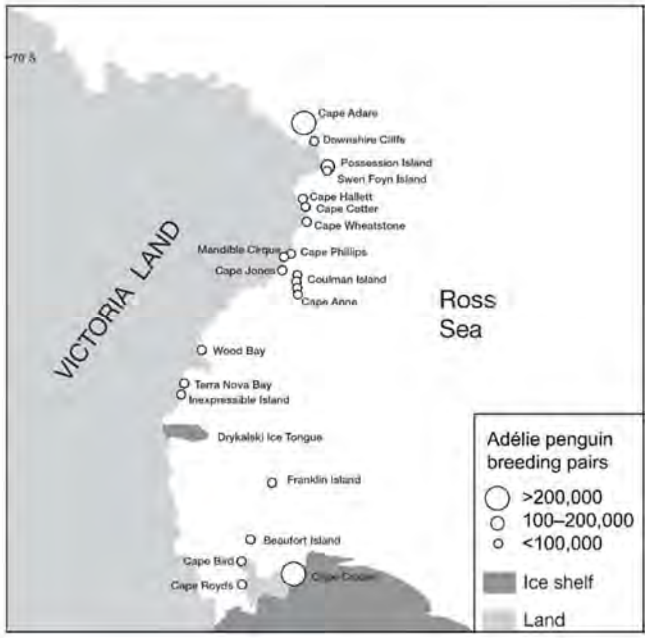


Figure 17.8: The number of breeding pairs of Adélie penguins in the Ross Sea from aerial census methods between 2001 and 2013 (Lyver, unpublished data).

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 seal (*Mirounga leonina*) are also present in the Ross Sea region but the nearest breeding colony is on Macquarie Island.

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 2010b), although minke whales are relatively common in summer.

Toothed whales present in the Ross Sea region include sperm whale (*Physeter macrocephalus*), killer whale (*Orcinus orca*), southern bottlenose whale (*Hyperoodon planifrons*), 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). Type-C (fish-eating) killer whale are considered to be by far the most common form in the McMurdo Sound region (extreme south-west of the Ross Sea), but the migration and feeding characteristics of this type are not well known.

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 17.9). 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.[9] Revisions and updates to the model are detailed in Pinkerton et al. (2016).

[9] 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.

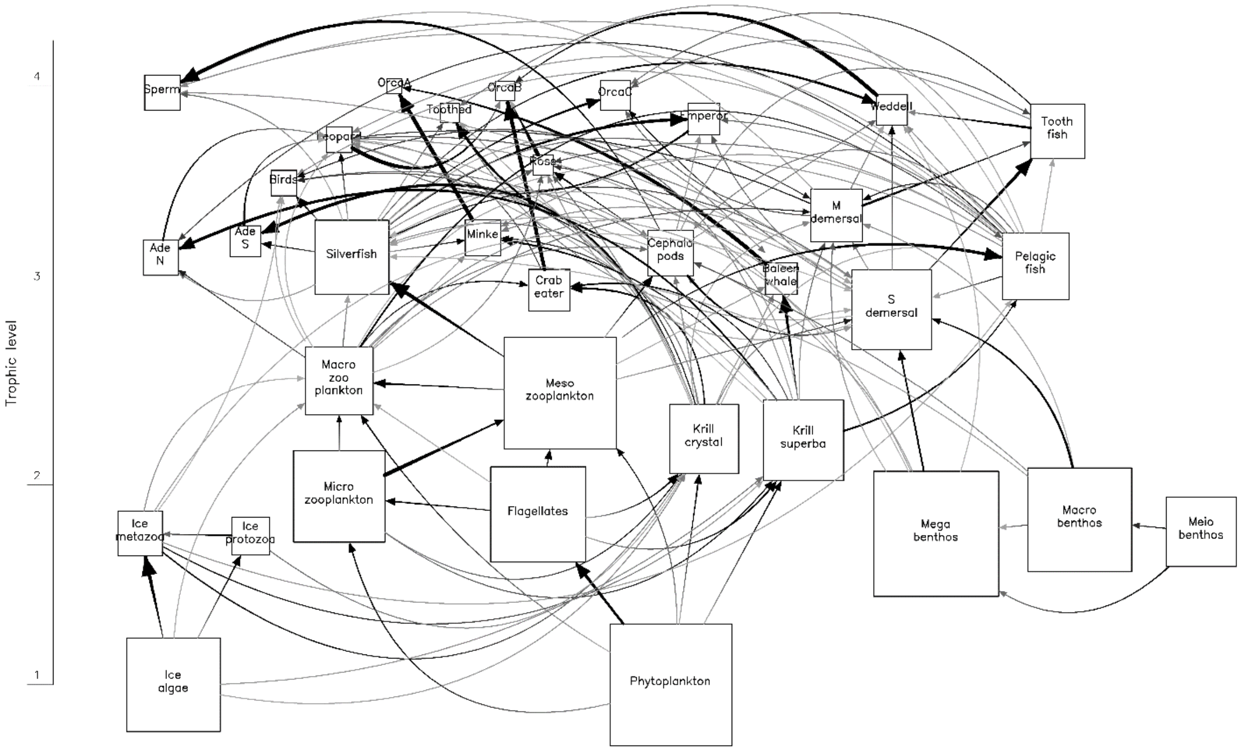


Figure 17.9: Ross Sea trophic model flow diagram, with arrows showing the direction of organic carbon flow. Bacterial and detrital groups omitted for clarity. Bigger boxes indicate more biomass. Boxes are positioned vertically according to trophic level. Thicker/darker lines show higher flows in or out 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 early polar expeditions killed an estimated 2000 Weddell seals to supply dog food in southern McMurdo Sound (Ainley 2009). The Weddell seal population in the southern Ross Sea was thought to have recovered from the mortality caused by polar expeditions by 1950 (Stirling 1971).

Blue, fin and sei whale 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 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 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). 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 (*Antimora rostrata*). A small bycatch of rock cods and ice cods is also taken.

Spatial distributions of fish bycatch are given in Figure 17.10.

The highest catch rates for macrourids, skates and eel cods are on the Ross Sea continental slope, in the area of the Iselin Bank. Icefish catch rates are higher on the Ross Sea slope and on the Amundsen Sea seamounts than elsewhere in the region. Deepsea (morid) cods have a more northern range and higher catch rates 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.

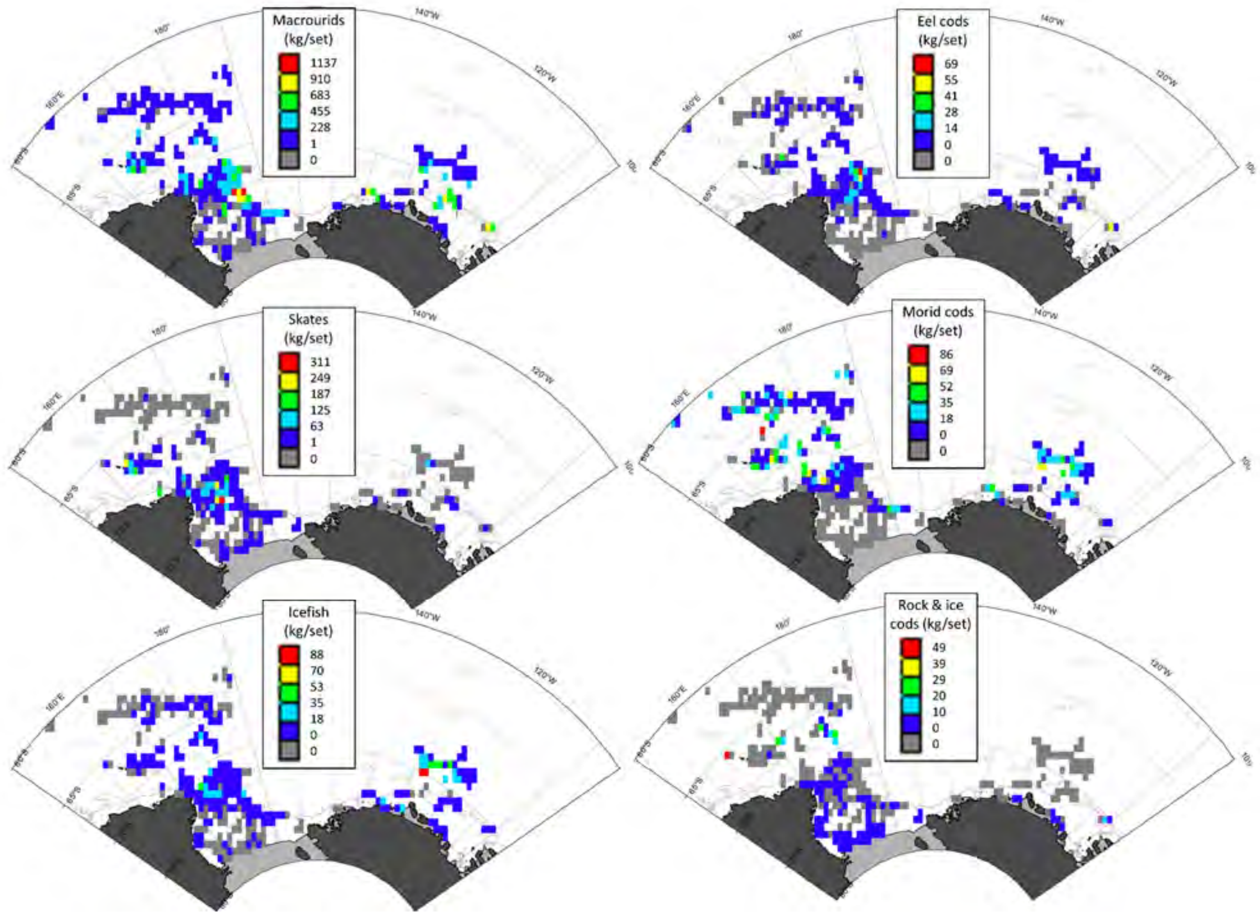


Figure 17.10: Catch rates of macrourids, skates, and icefish (left column); eel cods, morid cods, and rock cods & ice cods (right column). Depth contour at 1000 m. From Stevenson et al. (2012).

Macrourids

The main bycatch species in the Ross Sea are macrourids, which form 5–10% of the total catch per year or about 150 t/y (Large et al. 2015). 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.

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 based on the results of the IPY-CAMLR 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.

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). While 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. They form less than 1% of the catch brought onboard as most are cut-off alive at surface (Large et al. 2015). 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) and has never exceeded the bycatch limit for rajids (Large et al. 2015). Tagging of skates and rays as part of CCAMLR fisheries is not required at present.

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 depths. Catch rates are much lower than that 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). Although the survival rate of released skates in the Ross Sea region is unknown there have been 179 recoveries of tagged skates (Mormede & Dunn 2010). 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), probably related to method of identifying skates.

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 usually of the order of 10 t/y. Since 1998, a total of 11 species codes have been used for icefish caught in the Ross Sea fishery. Although most of the icefish catches were reported as 'unspecified icefish' (Stevenson et al. 2012), most of these are likely to be *C. dewitti* (Sutton et al. 2008).

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

Muraenolepsis ('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/hook. Overall catch and catch rate in Subarea 88.1 has been stable throughout the fishery, with one year of high catch (2007) when there were 19 sets reporting greater than 100 kg of *Muraenolepis* (Parker et al. 2012).

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). 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 *Muraenolepis* species suggest a relatively fast growing, semelparous,[10] 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, and 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, reaching 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) formed 0.01% of the reported catch between 1997 and 2012 (Stevenson et al. 2012). Annual average reported catch in this period was about 5 t, and most of this catch was taken from SSRUs 88.1G, 88.1L, and 88.1M (Figure 17.10). The highest catch rates for rock cods in this period were 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 (*Lepidonotothen kempi*), as 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*), as 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. 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

[10] 'Semelparous' means the adults breed once in their life then die.

(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 (*M. whitsoni*, n=227) and 62 years (*M. caml*, n=319). In contrast, *C. dewitti* and *M. evseenkoi* are faster-growing and shorter-lived species (maximum recorded age of 12 years). 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).

Modelling 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. 2014e). 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 includes 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. 2014e). This may affect the proportions of macrourids and icefish in the diet of toothfish over time (Mormede et al. 2014e).

17.3.4 EFFECTS ON PREDATOR SPECIES

Three species are known to predate 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; while 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 include 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. Recent information (Pitman, pers. comm., Torres et al. 2013) shows that type C killer whales in the Ross Sea can routinely dive to 200–400 m, with a maximum of over 700 m. 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 of killer whales and toothfish were analysed to test the hypothesis that toothfish were a major prey in the Ross Sea in summer but it was inconclusive.

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).

Recent evidence derived from 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 17.11; 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).

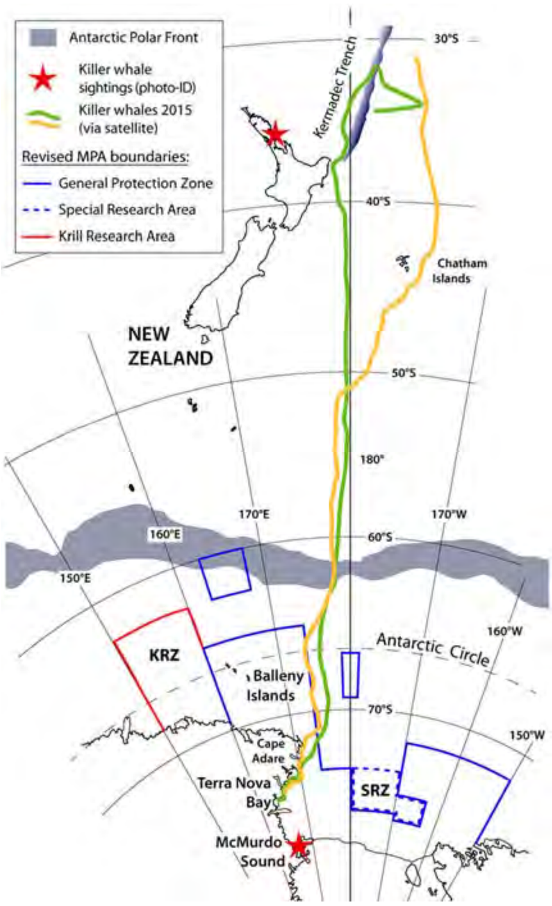


Figure 17.11: Satellite tracking of type C killer whales showing ecological connectivity for this species between the Ross Sea and New Zealand (Eisert et al. 2015).

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).

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 whales sightings in the area of 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, staying submerged for more than 1 h and commonly diving to about 400 m, reaching depths of 1185 m (Watkins et al. 1993, Perrin et al. 2002). Sperm whales in the Southern Ocean, and Pacific subantarctic 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 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 depths (about 1500 m) and occurrence around the Ross Sea slope, as well as photographic evidence (Eisert, pers. comm.) show that they also consume toothfish to some extent.

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 2015 was about 70%B_0[13] (Mormede et al. 2015). 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. 2014b), 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 through 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 importances were 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.[11]

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 sea floor 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^{o} latitude by 0.167^{o} 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 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 given the already low level of impact, that the improvement was very small.

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

[11] CCAMLR. VME Taxa Classification Guide 2009. Retrieved from https://www.ccamlr.org/en/system/files/VME-guide.pdf.

monitoring and management framework be able to detect this, and after how long?

Changes in age structure caused by changes in recruitment strength would likely be detected from the fisheries catch data themselves, but would not be apparent until the relevant cohort is 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 after 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. 2014d, Hanchet et al. 2015b, Dunn et al. 2016, Large et al. 2017). These surveys use a consistent, stratified design for sampling sub-adult toothfish in order 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 that changes on 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 in 20–30 years*.

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, 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 regions.

17.3.9 RESEARCH PRIORITIES

The medium-term (5–10 year) research priorities with regard to the Antarctic toothfish fishery in the Ross Sea and Amundsen Sea regions was 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 (MTR) 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* voyage to the Ross Sea in February 2015 included three days of a depth-stratified demersal trawl survey 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 in order 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 aging) (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014e).

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.[12] They describe that 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.[13] 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 MPA was approved by CCAMLR at its Commission meeting in October 2016. Conservation Measure 91-05 (2016) (CM 91-05)[14] details the specificities of the MPA. The boundaries of the MPA can be found in the CM or Figure 17.12. It has an area of 1.55 million km^2. It will to come 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 (2014).

The MPA will limit activities inside its boundaries in order to meet conservation, habitat protection, ecosystem monitoring, and fisheries management objectives (Tabl). The MPA will be divided into three zones:

- The General Protection Zone, which corresponds to 72% of the MPA, will be a '*no-take*' zone, which prohibits commercial fishing;
- The Special Research Zone (SRZ), which will permit some commercial fishing as a part of scientific research;
- The Krill Research Zone (KRZ), which will permit some harvesting of krill as a part of scientific research.

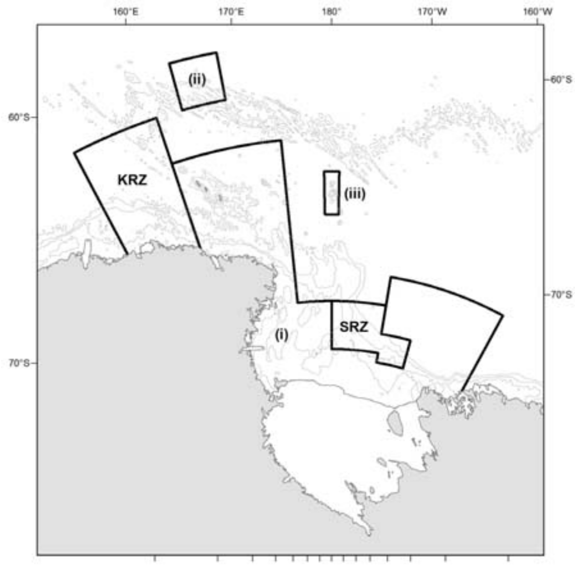


Figure 17.12: Map of the Marine Protected Area in the Ross Sea region. The black lines indicate the boundaries of the General Protection Zone (composed of areas (i), (ii), and (iii)), the Special Research Zone (SRZ), and the Krill Research Zone (KRZ). Depth contours are at 500 m, 1500 m and 2500 m.

Table 17.2: Objectives of the Marine Protected Area in the Ross Sea region. [Continued on next page]

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.

[12] CCAMLR. Marine Protected Areas. Retrieved from https://www.ccamlr.org/en/science/marine-protected-areas-mpas.

[13] Scientific Consensus Statement on Marine Reserves and Marine Protected Areas, https://www.nceas.ucsb.edu/Consensus.

[14] CCAMLR. Conservation Measure 91-05 (2016). Retrieved from https://www.ccamlr.org/en/measure-91-05-2016.

Table 17.2 [Continued]:

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.

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 the 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 the 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 the delivery of the research and monitoring plan.

There are no proposals to establish MPAs in the Amundsen 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 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-off from longlines (CM 33-03). This led to a fall in the number of rajids landed onboard and an increase in numbers released (Figure 17.13). There is no requirement to tag skates in the Ross Sea or Amundsen Sea regions at present.

Macrourid bycatch in the Ross Sea and Amundsen Sea regions increased to a maximum in 2005–06 as the fisheries expanded and then has decreased (Figure 17.14).

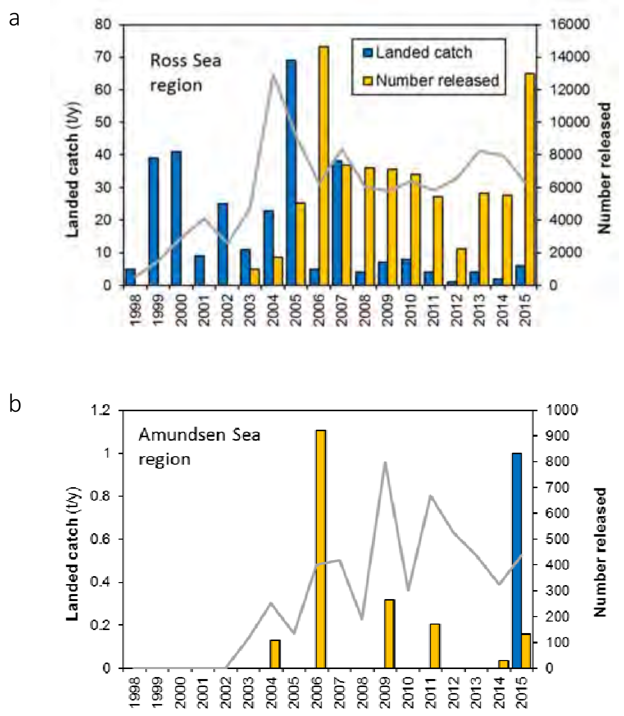


Figure 17.15: Catch of rajids (skates and rays). [a] Ross Sea region; [b] Amundsen Sea region. Weight of landed rajids (blue) and numbers cut off alive (orange). The relative effort (number of sets) is shown as the grey line (scale as in Figure 17.14).

Predation release of macrourids and icefish is expected to be larger than that fishing mortality, and may lead to increased abundance over time (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014e). 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 (Figure 17.15), 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 practice has 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 17.15 probably do not reflect changes in population sizes.

a

b

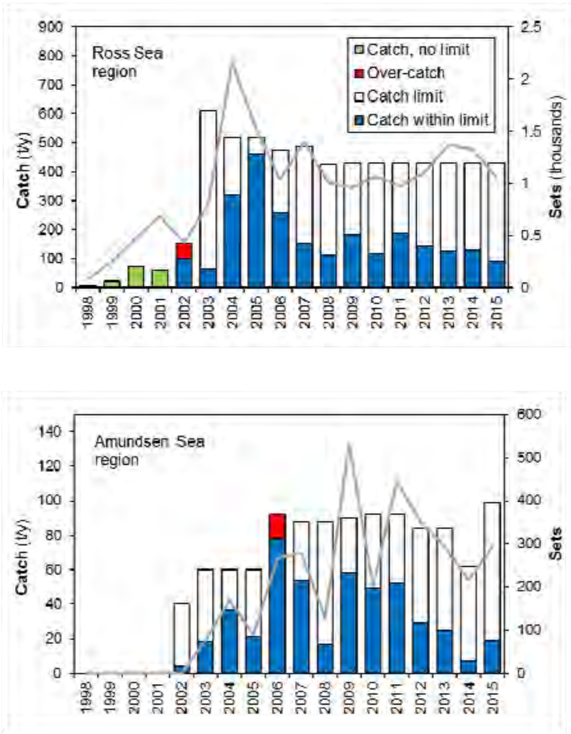
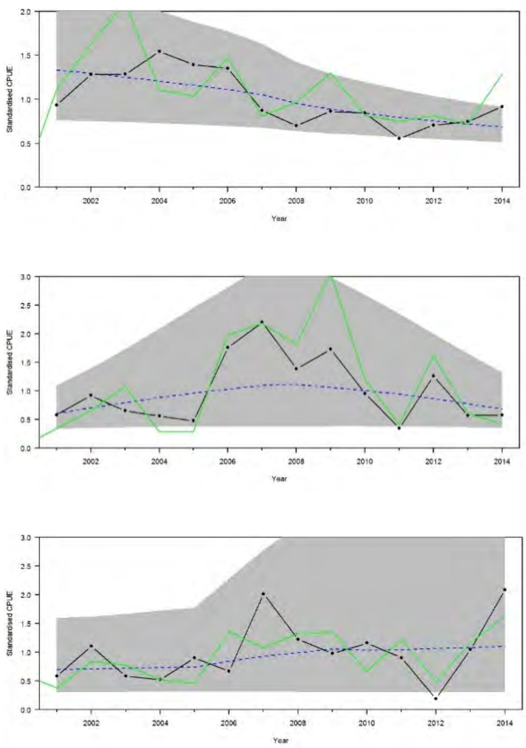


Figure 17.14: Macrourid bycatch (bars) and total fishing effort in terms of number of sets (grey line). [a] Ross Sea region; [b] Amundsen Sea region. White bars shows where the catch limit exceeds the catch, and red bars indicate that catch exceeded the catch limit in that year.

a

b

c



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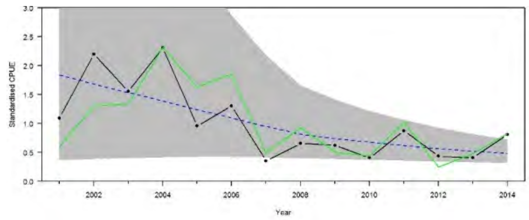

Figure 17.15: Raw (green), standardised (black), trends (blue) and 95% confidence intervals (grey) for catch-per-unit-effort (CPUE) in groups of bycatch species on the Ross Sea continental slope (small-scale research units (SSRUs) 88.1 H and I). [a] Macrourids; [b] icefish; [c] eel cods; [d] deepsea cod.

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 are being investigated.

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. 2014e). The differential strength of the predation release on macrourids and icefish would likely lead to a change in the diet of toothfish over time in favour of more icefish being consumed (Mormede et al. 2014e).

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 shore 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 come south of the Ross Sea slope.

Second, the size of toothfish consumed by predators is important as 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 predate on 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* (SPM; Mormede et al. 2014c). 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 to the length-frequency of toothfish taken over the Ross Sea shelf by the fishery are summarised in Large et al. (2015) and can be found in Figure 17.16.

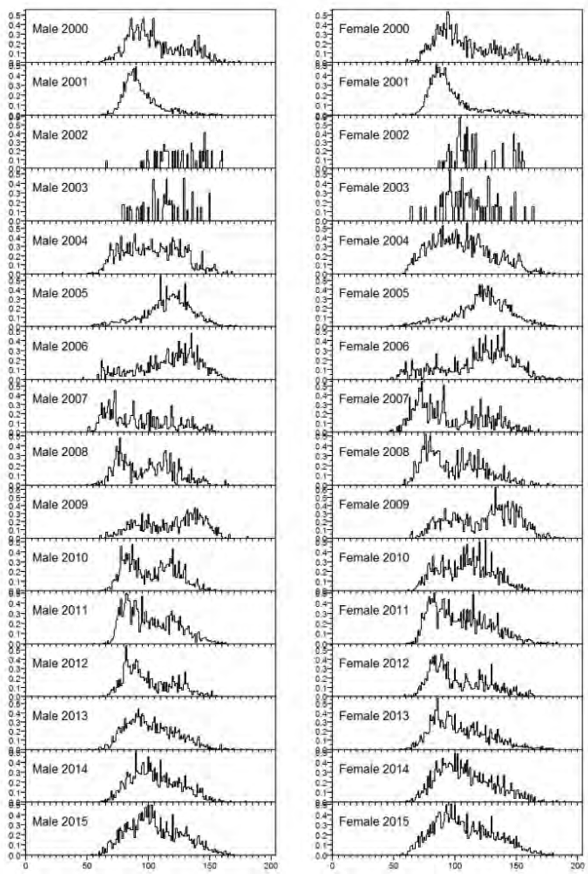

Figure 17.16: Estimated proportion of fish at length by sex for all vessels in the shelf region of the Ross Sea, for the years 2000–15 (Large et al. 2015).

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 17.17; Hanchet et al. 2015b) 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).

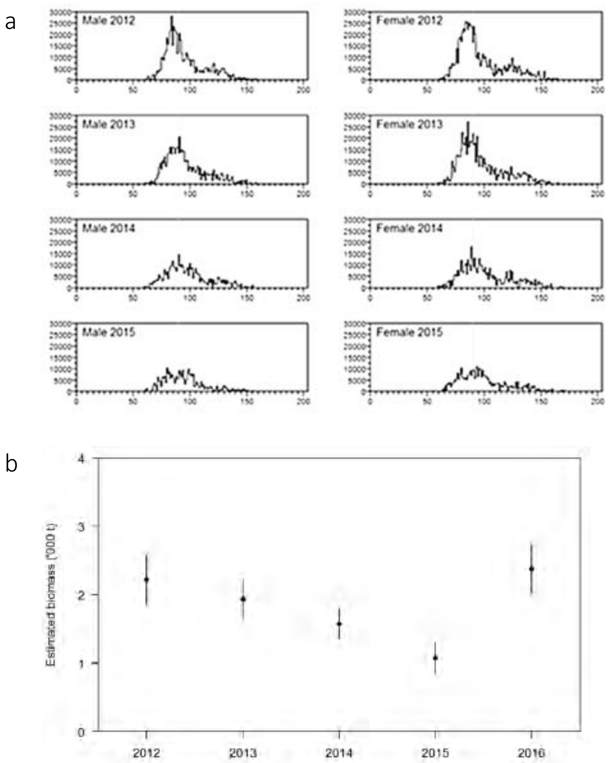

Figure 17.17: [a] Toothfish length frequency distributions in the Ross Sea shelf core strata (A–C) for the 2012–15 sub-adult surveys by sex (Hanchet et al. 2015b). [b] Comparison of standardised catch rates for the Ross Sea sub-adult surveys 2012–16. Error bars indicate the 95% confidence intervals (Dunn et al. 2016).

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 17.18). 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.

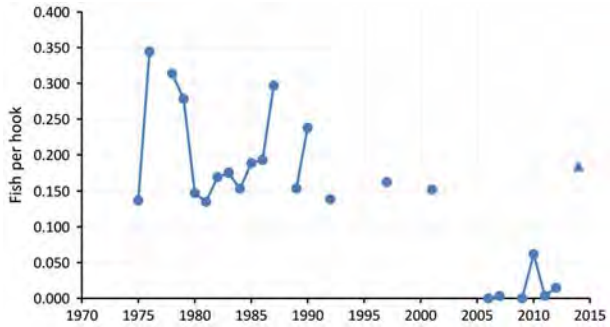

Figure 17.18: Catch rates (fish per hook) for toothfish sampled in McMurdo Sound, Antarctica, 1975–2014. Circles indicate pre-2013 data recalculated from Ainley et al. (2013) and triangle indicates the 2014 value from Parker et al. (2015). [Parker et al. (2015), figure 3].

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 (Figure 17.19a,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 lead 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 17.19c).

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).

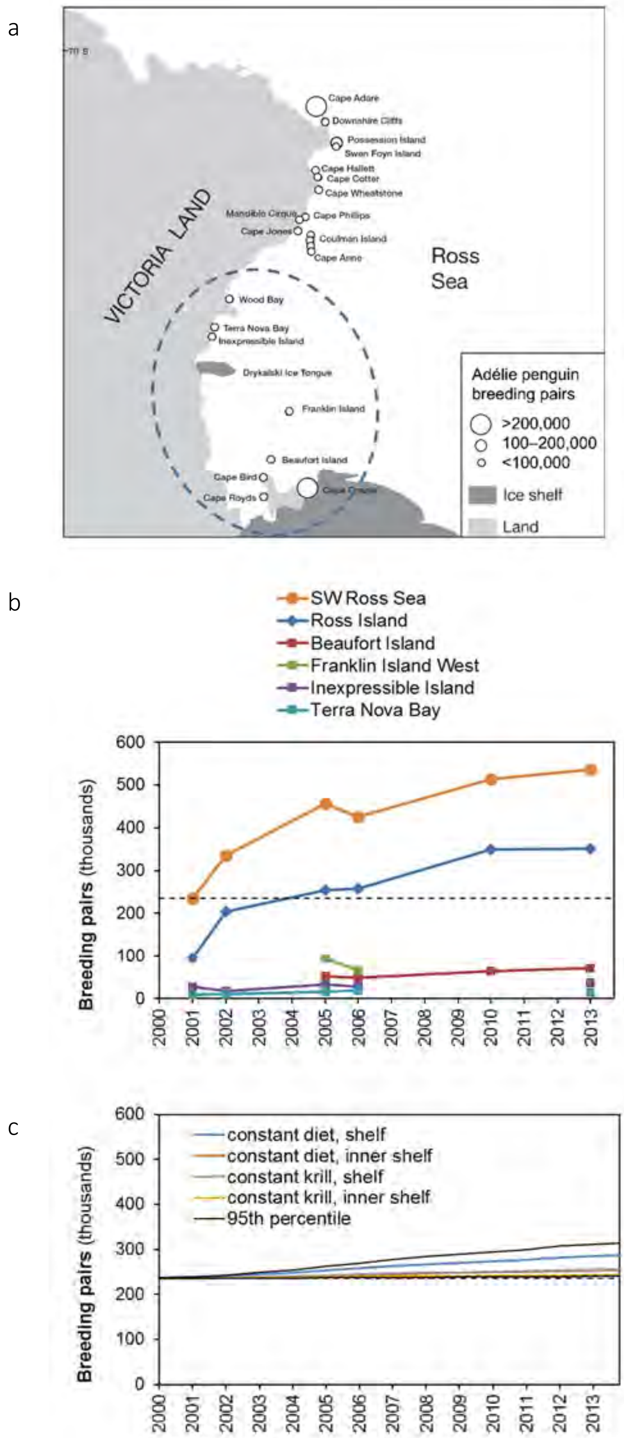

Figure 17.19: [a] Location and sizes of breeding colonies of Adélie penguins in the Ross Sea region. Those forming the 'southwest (SW) metapopulation' are enclosed in the dashed ellipse and forage over the Ross Sea shelf between chick hatching and fledging. [b] Changes to the total number of Adélie penguins 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 at the Ross Sea region, even with no specific management of VME impacts (Figure 17.20).

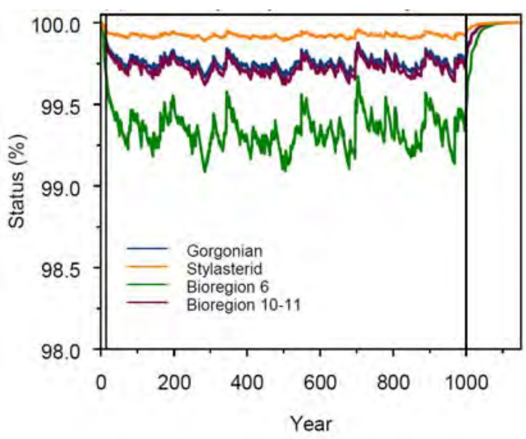

Figure 17.20: Simulated changes in the status of selected vulnerable marine ecosystems (VMEs) over time. Here, a status of 100% indicates that the habitat has the same extent and biomass as before fishing began and 0% indicates habitat removal at the scale of the Ross Sea region. Results are based on the medium-scale benthic habitat model of Dunn et al. (2010). The runs are for VMS 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

18. BIODIVERSITY

Scope of chapter	An overview of the MPI Marine Biodiversity Programme, describing the national and global context for marine biodiversity research in New Zealand; summarises the research findings and progress of the MPI Biodiversity Research Programme from 2000–17, including one-off, whole-of-government research initiatives administered under this programme (e.g., Ocean Survey 20/20 Biodiversity and Fisheries projects; International Polar Year Census of Antarctic Marine Life project, Strategic Science Investment Fund project on quantifying predictive habitat modelling).
Area	New Zealand Territorial Seas, EEZ and Continental shelf extension (BioInfo); South-west Pacific Region associated with South Pacific Regional Fisheries Management Organisation (SPRFMO); Southern Ocean and Ross Sea region (BioRoss)
Focal issues	• New Zealand seas have globally significant levels of endemic marine biodiversity, particularly in coastal habitats, offshore island habitats and on underwater topographical features like seamounts and canyons. • Mapping and documenting the identity, abundance and distribution patterns of New Zealand's marine biodiversity in this extremely large area of responsibility (~5.8 million km^2) is far from complete. Identifying areas of high biodiversity remains a challenge, particularly for environmental impact evaluation or assessing response to climate change scenarios. • The state of marine biodiversity in New Zealand (and whether or not it is declining) is not reported nationally. New Zealand is still in the characterisation stage of recording and plotting the distribution of species, and the discovery of new species has not begun to level off yet. • Selecting a suite of suitable ecosystem indicators and monitoring of marine biodiversity remains challenging. In addition to extinctions or reduction in species richness and abundance, proxies such as environmental degradation, species invasion and hybridisations, habitats that have been diminished or removed, and the disruption of ecosystem structure and function, as well as ecological processes (e.g., biological cycling of water, nutrients and energy) could be used as indicators. • The efficacy of current spatial measures and management actions to protect sea life in achieving the goal of the NZ Biodiversity Strategy 'Halting the decline of biodiversity' across all of New Zealand's marine environment is not known due to inadequate sampling effort. Good quality data exist for some spatial measures, but scaling up remains challenging. • Climate change effects on the ocean are occurring in New Zealand, and as these continue, the likely consequences for marine biodiversity and productivity are significant. • Marine biodiversity has not become an integral part of business or strategic planning across relevant government agencies. The functional role of marine biodiversity and ecosystem services in providing a healthy ecosystem, maintaining environmental limits and maintaining sustainability are not well recognised.
Key progress 2015–16	• Six new projects began in the 2014–15 financial year: ZBD2014-01 *Live DW coral experiment phase 2*; ZBD2014-04 *Isoscapes for trophic studies*; ZBD2014-05 *Ocean acidification modelling*; and ZBD2013-08 *NZ-Ross sea connectivity Humpback whales* have been completed.

	ZBD2014-10 *BPA biodiversity*; ZBD2014-06 *Macroalgae mapping and potential as national scale indicators*; ZBD2014-07 *Southern coralline algae shellfish habitat*; and ZBD2014-09 *Climate change risks and opportunities* are progressing and nearing completion. A further three projects were initiated in 2015–16: ZBD2016-07 *Multiple Stressors on Coastal Ecosystems-in situ*; ZBD2016-11 *Quantifying Benthic biodiversity Across natural Gradients*; and ZBD2016-04 *Organic Carbon Recycling*.
Emerging issues and ongoing gaps	• The combined effects of multiple stressors arising from climate change and a range of other anthropogenic activities on biodiversity and marine ecosystems (structure and function) are likely to be large and complex. MPI is working with the Sustainable Seas National Science Challenge to address cumulative effects arising from multiple use. • Ecosystem approaches to marine resource management are urgently needed, particularly with the renewed interest in developing the marine economy through marine mining and extraction activities. There is increasing pressure for MPI to adopt an EBFM framework, for example in the New Zealand Biodiversity Action Plan 2016 and, the feedback from the public consultation on the Future of Our Fisheries, sought in 2016. • The nature and functional role of marine microbial biodiversity in large-scale biogeochemical and ecosystem processes may be crucial to productivity in our seas, but are not well understood. • Genetic and life-history stage connectivity between and within large-scale habitats are likely to be important to the size and placement of marine protection zones and to their success. • Long-term observations (e.g., decadal to millennia timeframes) of variability and change in the marine environment (including biodiversity) are not yet generally available at geographic scales appropriate for national reporting. • Metrics for assessing the effectiveness of current protection measures in safeguarding marine biodiversity and aquatic ecosystem health in New Zealand and the Ross Sea region are inadequate. • Economic value of ecosystem goods and services provided by marine biodiversity to current and future generations are not yet addressed in extractive business models. • Conservation of marine biodiversity, monitoring, reduction of loss and enhancement are emerging requirements for signatories (including New Zealand) to the CBD Aichi-Nagoya Agreement 2010. In October 2016, New Zealand released the updated NZ Biodiversity Action Plan 2016–2020, which identifies some goals that are specific to fisheries and sustainable practices. • Geo-engineering methods including ocean fertilisation continues to be advocated in some areas of international climate change mitigation. • Meeting New Zealand international responsibilities includes participation in international data collection programmes and long-term commitment, e.g., SAHFOS, IMOS, SOCPR ARGO and BIO-ARGO. • Biodiversity indicators that can be used to evaluate the health of the marine ecosystem and biodiversity loss need to be consolidated. • Marine debris and pollution are increasing in New Zealand waters, particularly in the coastal zone, but are not adequately addressed at present. • The effects of land-use practices on coastal ecosystems are still not fully addressed by management agencies (see Chapter 15).

	• Validation data to support habitat suitability models is inadequate for most species in most of the EEZ, therefore there is a need for more robust methods that can work with limited data. • The Kaikoura earthquake November 2016 has provided the unique opportunity to document the effects of uplift on fisheries and the marine ecology of the area.
MPI and cross-government research (current)	74 biodiversity projects have been commissioned over the period 2000–16. The five-year programme initiated in 2011 addressed seven science objectives in the Biodiversity Programme: 1. characterisation and description; 2. ecosystem scale biodiversity; 3. functional role of biodiversity; 4. genetics; 5. ocean climate effects; 6. indicators; 7. threats to biodiversity. During this period, the MPI Biodiversity Research Advisory Group has had strong synergies with fisheries research funded by the MPI Aquatic and Environment Working Group (AEWG). As MPI develops the Future of Our Fisheries programme, and addresses commitments under the New Zealand Biodiversity Action Plan 2016, it is likely that BRAG will revise its objectives to better align with Ecosystem Based Fisheries Management information needs.
NZ biodiversity research (research institutes, universities current)	Research programmes and database initiatives on Marine Biodiversity are run at University of Auckland (World Register of Marine Species (WoRMS), marine reserves, rocky reef ecology, soft sediment functional ecology, Ross Sea meroplankton, genetics); Auckland University of Technology, University of Waikato (soft sediment functional ecology and biodiversity), Victoria University of Wellington (monitoring marine reserves, population genetics), University of Canterbury (intertidal and subtidal ecology, kelp forests and biodiversity), University of Otago (land-use effects, bryozoans, inshore ecology, ocean acidification), National Institute of Water and Atmospheric Research (NIWA) and Cawthron Institute. Relevant Marine Platform programmes at NIWA include Coasts and Oceans Centre programmes and the NIWA Fisheries Centre. DOC, MPI, NIWA and Landcare Research contribute to the NZ Organisms Register. MBIE National Science Challenges 'Deep South' and 'Sustainable Seas' are hosted by NIWA. Other relevant research includes the large MBIE funded project CARIM (Coastal Acidification: Rates, Impact and Management), which BRAG is co-funding with two small fisheries related projects.
Links to MPI objectives	As the new review, Future of Our Fisheries, develops into a revised work programme, BRAG alignment with EBFM objectives will likely strengthen.
Links across government	The Biodiversity Research Advisory Group engages in Natural Resource Sector discussions (MfE, DOC, MBIE, LINZ, EPA, MOT, Maritime NZ, Antarctica NZ, NZ Statistics, MFAT) and whole-of-government projects such as Ocean Survey 20/20, International Polar Year, identification of strategic research needs in marine science, the National Science Challenges (MBIE, NIWA), environmental reporting (MfE) and the NZ Biodiversity Action Plan (DOC).
Related chapters/issues	Multiple use of marine resources, land-based effects, variability and change, marine monitoring, cumulative effects of use and extraction in the marine environment, protected areas, benthic impacts, ecosystem approaches to fisheries and marine resource management.

Note: Only Sections 18.1, 18.2 and 18.5 of this chapter has been updated for the AEBAR December 2017.

18.1 INTRODUCTION

This chapter summarises the development and progress of the MPI Biodiversity Research Programme 2000–17, and reviews the work commissioned in the context of national and global concerns about biodiversity and the maintenance of the marine ecosystem in a healthy functioning state, as identified by the New Zealand Biodiversity Strategy (NZBS; Anon 2000).

18.1.1 HALTING THE DECLINE IN BIODIVERSITY

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. The Desired Outcomes by 2020 for the marine environment (Coasts and Oceans, Theme 3) in the NZBS were stated as:

- 'New Zealand's natural marine habitats and ecosystems are maintained in a healthy functioning state and degraded marine habitats are recovering.
- A full range of marine habitats and ecosystems representative of New Zealand's indigenous marine biodiversity is protected.
- No human-induced extinctions of marine species within New Zealand's marine environment have occurred.
- Rare or threatened marine species are adequately protected from harvesting and other human threats, enabling them to recover.
- Marine biodiversity is appreciated, and any harvesting or marine development is done in an informed, controlled and ecologically sustainable manner.'

In the marine environment, biodiversity decline is characterised not only by extinctions or reduction in species richness and abundance, but also by environmental degradation such as species invasion and hybridisations, habitats that have been diminished or removed, and the disruption of ecosystem structure and function, as well as ecological processes (e.g., biological cycling of water, nutrients and energy). Measuring the decline of marine biodiversity is complicated by the 'shifting baseline syndrome', a common obstacle to useful biodiversity assessment and monitoring.[1] Furthermore the size range of organisms sampled is often limited to macroscopic. Changes (declines) in biodiversity metrics at a macroscopic level may not detect potentially large changes in biodiversity in smaller-sized organisms below our sampling threshold that may also be critical to marine ecosystem health and well-being.

Responsibility for implementing New Zealand's Biodiversity Strategy is led by the Department of Conservation (DOC), with significant input from the Ministry for Environment (MfE), and the Ministry of Fisheries (now part of MPI).[2] In 2016 DOC completed a process to refresh the Biodiversity Strategy to better meet the Aichi Agreement, and has released New Zealand's Biodiversity Action Plan 2016–2020.[3]

There are several new goals in the Action Plan that are of direct relevance to MPI:

Goal B: National Target 5. Biodiversity is integrated into New Zealand's fisheries management system

Goal D: National Target 12. More Threatened, At Risk, or Declining species are managed to the extent necessary to

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[1] Australian Government. A National Approach to Addressing Marine Biodiversity Decline. Retrieved from Uwww.environment.gov.au/coasts/publications/marine-diversity-decline/index.html.

[2] Department of Conservation (DOC). Biodiversity. Retrieved from https://www.biodiversity.govt.nz/picture/doing/programmes/index.html.

[3] DOC. New Zealand Biodiversity Action Plan 2016–2020. Retrieved from http://www.doc.govt.nz/Documents/conservation/new-zealand-biodiversity-action-plan-2016-2020.pdf.

minimise extinction risk and ensure genetic diversity is maintained

Goal E: National Target 17. Whānau, hapū and iwi are better able to practice their responsibilities as kaitiaki.

18.1.2 DEFINING BIODIVERSITY

New Zealand's Biodiversity Strategy defines biodiversity as:

'The variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part [as defined by the CBD]; this includes diversity within species, between species and of ecosystems [as further disaggregated for New Zealand purposes]. Components include:

- *Genetic diversity: the variability in the genetic make-up among individuals within a single species. In more technical terms, it is the genetic differences among populations of a single species and those among individuals within a population.*
- *Species diversity – the variety of species – whether wild or domesticated – within a particular geographic area.*
- *Ecological diversity – the variety of ecosystem types (such as forests, deserts, grasslands, streams, lakes wetlands and oceans) and their biological communities that interact with one another and their non-living environments.'*

MPI's Biodiversity programme is concerned primarily with research to underpin NZBS Theme 3: Biodiversity in Coastal and Marine Ecosystems:

'Coastal and marine ecosystems include estuaries, inshore coastal areas and offshore areas, and all the resident and migratory marine species that live in them.'

New Zealand's ocean territory (including territorial sea and the recent continental shelf extension) [4] is very large relative to the area of land,[5] and includes some 15–18 000 kilometres of coastline extending from the subtropical north to the cool subantarctic waters in the south. New Zealand also has a rich marine biodiversity that has been recognised as being globally significant with up to 38% of all marine species (46% for Animalia) estimated as endemic (Gordon 2013) and comprising up to 8% of global marine biodiversity. [These statistics do not include estimated undiscovered species, which are likely to increase the proportion of endemics.]

About 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. Several factors would suggest that this estimate of total marine species is conservative. Such factors include the region's size, the depth range, geomorphological and hydrological complexity, 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 like myxozoans and nematodes, as well as free-living nematodes. If recent indications of massive oceanic microbial diversity are also taken into account (e.g., Sogin et al. 2006) then the number above is certainly conservative, perhaps indeterminable.

New Zealand's marine biodiversity is affected by many uses of the marine environment, particularly fishing, aquaculture, shipping, petroleum and mineral extraction, renewable energy, tourism and recreation.[6] Impacts from

[4] New Zealand Foreign Affairs and Trade. Our maritime zones and boundaries. Retrieved from https://www.mfat.govt.nz/en/environment/oceans/our-maritime-zones-and-boundaries.

[5] New Zealand sea area is about 5.8 million km^2 including TS, EEZ and continental shelf extension; the fourth largest in the world; www.linz.govt.nz.

[6] Royal Society of New Zealand (RSNZ). Future Marine Resource Use. Retrieved from http://www.marinenz.org.nz/documents/Future-Marine-Resource-Use-web.pdf; Statistics NZ. Fish. Retrieved from http://www.stats.govt.nz/browse_for_stats/environment/natural_resources/fish.aspx.

changing land use, including agricultural, urban run-off and coastal development can also affect marine biodiversity (Morrison et al. 2009). 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 – see New Zealand Royal Society Workshop papers). The growing arrival of non-indigenous (sometimes invasive) marine species is also a threat to local biodiversity (e.g., Coutts & Dodgshun 2007, Cranfield et al. 2003, Gould & Ahyong 2008, Russell et al. 2008, Williams et al. 2008).

Understanding about 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). Current 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). However, New Zealand is less well placed when it comes to understanding the threats to marine biodiversity (Costello et al. 2010, MacDiarmid et al. 2012) and the nature of their impacts. Marine invasion and the effects of climate change and acidification of the ocean are key threats, and anthropogenic threats from increasing resource use, high levels of agricultural runoff and sedimentation as well as marine debris are causing localised degradation of marine habitats (Kingsford et al. 2009, Lundquist et al. 2011).

There are ongoing concerns about the decline of some key species (MfE 2016[7]), 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) despite the combined efforts of New Zealand's government and stakeholders to fill information gaps and reduce impacts on biodiversity. 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).

New Zealanders increasingly value environmental, economic and social aspects of marine biodiversity and the ecosystem services that a healthy marine environment provides. They also value the need to sustainably manage the use of coastal and marine environments and maintain biological diversity as reflected by recent policy statements by the New Zealand government.[8,9] 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. Trying to manage this in a more holistic way through EBFM has already been identified by MPI as a medium-term goal in the Future of Our Fisheries review.[10]

The government's Business Growth Agenda acknowledges the lack of progress in halting the decline in biodiversity in New Zealand, and indicates that development of the marine economy requires a careful approach to the environment. Many documents have been written about the lack of basic ecological characterisation throughout the Territorial Sea and EEZ (Snelder et al. 2006, Leathwick et al. 2010, Bowden et al. 2011a) and a fully consulted programme to 'map the farm' through Ocean Survey 2020 was initiated in 2006. Funding for the programme became scarce in 2012 and effectively stopped. No priority or action

[7] Ministry for the Environment (MfE). Our marine environment 2016: Data to 2015. Retrieved from http://www.mfe.govt.nz/sites/default/files/media/Environmental%20reporting/our-marine-environment.pdf.

[8] MfE. Proposed National Policy Statement on Indigenous Biological Diversity (biodiversity) under the Resource Management Act 1991. Retrieved from http://www.mfe.govt.nz/publications/land-biodiversity/proposed-national-policy-statement-indigenous-biodiversity.

[9] DOC. New Zealand Coastal Policy Statement 2010. Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/coastal-management/nz-coastal-policy-statement-2010.pdf.

[10] Ministry for Primary Industries (MPI). Future of our Fisheries. https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/strengthening-fisheries-management/future-of-our-fisheries.

point has been identified to rectify the ongoing knowledge gap.

The most recent introduction of new legislation is the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012. However, progress on an integrated oceans policy and strategic direction for implementation of New Zealand's Biodiversity Strategy has not been as rapid in New Zealand compared with other countries such as Canada, the UK, the USA and Australia (Peart et al. 2011).

18.1.3 IMPLEMENTATION OF NEW ZEALAND'S BIODIVERSITY STRATEGY

A number of initiatives have been supported by MPI to meet the goals of the NZBS. Commitments include the creation of NABIS (the National Aquatic Biodiversity Information System), [11] the continuation of the MPI Biodiversity Research Programme, convening and chairing the Biodiversity Research Advisory Group, and working in the Marine Protected Area policy arena with DOC and MfE. [12] DOC surveys and monitors aspects of marine biodiversity, particularly in marine reserves and in relation to protected and threatened marine species. [13] New Zealand's no-take marine reserves have particular advantages over the other 94% of global marine-protected areas that allow fishing (Costello & Ballantine 2015) in that monitoring of any changes (including improvements) is not confounded by the effects of species extractions.

MfE has encouraged Regional Councils to develop coastal monitoring programmes, and with MPI and DOC initiated an approach to Marine Environmental Classification. Biodiversity-related research has also been carried out through MPI's Biosecurity Science Strategy. One result included mapping and valuation of marine biodiversity around New Zealand's coastline. Periodic marine 'BioBlitzes' around New Zealand yield surprising numbers of new species, even in presumed well-studied areas, have the advantage of engaging children, parents and teachers in discovery, and involve research scientists who go on to 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.[14]

Marine biodiversity research 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,[15] as well as to local programmes that have improved understanding of the role of biodiversity in the marine ecosystem.

The museums of Auckland, Canterbury, Otago and the Museum of New Zealand (Te Papa) also conduct biodiversity sampling expeditions and national collections of specimens have been set up within museums and at NIWA. Regional Councils give effect to NZBS, NZ Coastal Policy Statement 2011, and spatial planning. The Hauraki Gulf Forum and the marine spatial planning tool 'Seachange' run by the Auckland and Waikato Regional Councils have taken a holistic and integrated approach to marine management and restoring the marine productivity in the area.[16]

18.1.4 CURRENT CHALLENGES AND AGENDAS

Since the launch of the Biodiversity Strategy, there have been substantial changes in government policy and changes to the science agenda in New Zealand. Improving New Zealand's economic performance while continuing to strengthen society and protect the environment has been a key component for the marine economy as the general

[11] NABIS is an interactive database, accessible at www.nabis.govt.nz.
[12] MfE. Marine protected areas. Retrieved from http://www.mfe.govt.nz/marine/reforms/marine-protected-areas.
[13] DOC, www.doc.govt.nz.
[14] Curious Minds. Our goals and actions. Retrieved from http://www.curiousminds.nz/actions.
[15] National Ocean Service. What is the Census of Marine Life. Retrieved from https://oceanservice.noaa.gov/facts/marine-census.html.
[16] Sea Change: Hauraki Gulf Marine Spatial Plan. Retrieved from http://www.seachange.org.nz.

public have become increasingly aware of how human behaviour and actions affect the environment.

New Zealand's Business Growth Agenda (BGA) was updated in 2015, and the Natural Resources Chapter mentions biodiversity under Goal 6:[17]

> '*Develop our aquaculture, fisheries and other marine resources, while maintaining marine biodiversity and sustainability.* '

The latest initiative from the government is the '*Conservation and Environment Science Roadmap that identifies the areas of scientific knowledge needed by government over the next 20 years to support decision-making for conservation and environmental policy and management to achieve the most desirable future for New Zealand*'. [18]

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, minerals (Business Growth Agenda 2014). 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).

18.1.5 FUTURE DIRECTIONS

At the time of writing, the Ministry of Business, Innovation and Employment state the following in the National Statement of Scientific Investment:

> '*Effective environmental management can underpin economic goals. There is a significant opportunity for improving New Zealand's environmental management to improve our information and evidence base, and our understanding of environmental opportunities and limits. Unique natural resources require a New Zealand-specific approach Government-funded environmental research should be relevant to the local context and natural heritage. It should take into account government's wider strategic plans and priorities for environmental management, and the unique relationship of Māori with their environment. For example, priority research might support national and international environmental commitments, or the goals of the Crown-Māori economic growth partnership He kai kei aku ringa. We will balance productivity and sustainability. Where environmental research has strong links to primary industries, we will continue to seek an appropriate balance of public and private investment. We will seek to fund research that increases the productive potential of our environment, while preserving and enhancing its quality and sustainability*'.[19]

- Develop our aquaculture, fisheries and other marine resources, while maintaining marine biodiversity and sustainability, BGA Refresh 2017.[20]
- Develop guidance on biodiversity offsetting, to assist businesses to achieve economic and environmental objectives, DOC.
- Grow the number of new business opportunities on public conservation land, such as species tourism, in order to deliver increased economic prosperity and conservation gain, DOC.

[17] Ministry for Business Innovation and Employment (MBIE). Building Natural Resources. Retrieved from http://www.mbie.govt.nz/info-services/business/business-growth-agenda/pdf-and-image-library/towards-2025/BGA%20Natural%20Resources%20Chapter.pdf.

[18] MfE. Conservation and Environment Science Roadmap. Retrieved from http://www.mfe.govt.nz/about-us/our-policy-and-evidence-focus/conservation-and-environment-science-roadmap.

[19] MBIE. National Statement of Science Investment 2015–2025. Retrieved from http://www.mbie.govt.nz/info-services/science-innovation/pdf-library/NSSI%20Final%20Document%202015.pdf.

[20] MBIE. Building Natural Resources. Retrieved from http://www.mbie.govt.nz/info-services/business/business-growth-agenda/pdf-and-image-library/2017-documents/building-natural-resources.pdf.

- Engage with local councils to improve coordination and more effectively manage biodiversity and ecosystem services, DOC.
- Establish a Forum with Māori and the private sector to discuss opportunities related to natural resources, MPI, MBIE, TPK.
- Partner with Māori, and other primary industry participants, to enable initiatives to advance the productivity of Māori agribusinesses, MPI.
- Investigate the development of an investment fund for commercial discovery processes from Māori collectively owned assets, MBIE, MPI, TPK.

18.1.6 FISHERIES AND THE ENVIRONMENT

Most of New Zealand's commercial fisheries are wild-caught, and continuity of their productivity is dependent on the retention of a healthy functioning marine ecosystem. The 'license to operate' is mandated by the Fisheries Act 1996 that 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.

The marketplace is continually evolving and as social awareness of commercial activity in the sea has increased across the globe, the 'social license' or 'acceptance' of commercial fishing activities has come under higher scrutiny in New Zealand. Industry and government have gone beyond legislative requirements to maintain stocks at healthy levels and to demonstrate stewardship of the natural environment as well as the fishstocks; eco-labelling certification such as the Marine Stewardship Council (MSC) of some key fisheries is just one manifestation of this. It will become increasingly important to seek social license to operate and to demonstrate responsible environmental stewardship in other industries as New Zealand stretches towards broader goals of growing the marine economy. Developing ecosystem based management (EBM) and understanding social license to operate in the marine environment are central to the government's National Science Challenge 'Sustainable Seas'.[21]

The large-scale threats to the marine environment and biodiversity posed by increasing global changes such as 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, hydrocarbon and mineral extraction) are increasingly being recognised in policy and government circles (e.g., Office of the Prime Minister's Science Advisory Committee 2013,[22] NZ Royal Society 2016,[23] Statistics New Zealand 2013, Statistics New Zealand 2012, Royal Society 2009, Capson and Guinotte 2014; MBIE funded research programmes[24]).

Despite this recognition, progress on tackling marine-related climate change effects, investment in long-term monitoring of the marine environment and ready access to data remains slow. Long-term monitoring and environmental reporting has however been recognised as a major gap by the government and in September 2015 the Environmental Reporting Act 2015 was passed into law.[25] The new framework for environmental reporting divides the environment into five environmental domains. Under

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[21] MBIE. Sustainable Seas. Retrieved from http://www.mbie.govt.nz/info-services/science-innovation/national-science-challenges/sustainable-seas.

[22] Office of the Prime Minister's Science Advisory Committee (2013) New Zealand's changing climate and oceans: The impact of human activity and implications for the future. Retrieved from http://www.pmcsa.org.nz/wp-content/uploads/New-Zealands-Changing-Climate-and-Oceans-report.pdf.

[23] Royal Society of New Zealand. New Zealand vulnerable to the threats of climate change – report finds. 19 April 2016. Retrieved from https://royalsociety.org.nz/news/new-zealand-vulnerable-to-the-threats-of-climate-change-report-finds.

[24] MBIE. National Science Challenges. Retrieved from http://www.mbie.govt.nz/info-services/science-innovation/national-science-challenges.

[25] MfE. About the Environmental Reporting Act 2015. Retrieved from http://www.mfe.govt.nz/more/environmental-reporting/about-act.

each domain three main types of information are reported on: pressures, states and impacts.

Topics for each domain have been identified and set in the Environmental Reporting (topics for Environmental Reports) Regulations 2016 under the Environmental Reporting Act 2015. 'Our marine environment 2016' is the first report released under the new legislation.

Understanding about the rate of change and the resilience of biodiversity in response to the cumulative effects of multiple stressors across large spatial scales (e.g., ocean acidification, temperature increase and oxygen depletion) remain an ongoing topic of investigation. 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., Blasiak et al. 2017; Phillips & Perez-Ramirez 2017), including New Zealand (e.g., Coastal Acidification: Rates, Impacts and Management.[26]

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 the 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).

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, 2014a, National Plan of Action (NPOA) - Seabirds 2013, NPOA Sharks 2013) and some of these have been followed up with a Spatially Explicit Risk Assessment.[27]

18.2 GLOBAL UNDERSTANDING AND DEVELOPMENTS

Worldwide, there is concern about biodiversity and the current rate of decline. The current doctrine states that

- Greater species diversity ensures natural sustainability for all life forms
- Healthy ecosystems can better withstand and recover from impacts

In addition, the message that biodiversity provides a number of natural services for everyone, such as ecosystem services, biological resources and social benefits, is slowly reaching politicians and industry as people ask, 'Why is biodiversity important? Does it really matter if there aren't so many species?' [28] It has now been shown that biodiversity boosts ecosystem productivity where each species, no matter how small, has an important role to play. It has also been shown that greater species diversity underpins system resilience and natural sustainability for all life forms; and healthy ecosystems can better withstand and recover from a variety of disasters (Hughes et al. 2005, Thrush et al. 2009).

18.2.1 THE DECADE OF BIODIVERSITY 2011–2020

At its 65th session, The United Nations General Assembly declared the period 2011–2020 to be '*the United Nations Decade on Biodiversity, with a view to contributing to the implementation of the Strategic Plan for Biodiversity for the period 2011–2020*' (Resolution 65/161). The decade serves to support and promote implementation of the objectives of the Strategic Plan for Biodiversity and the Aichi-Nagoya Biodiversity Targets. The principal instruments for

[26] CARIM, http://www.carim.nz.

[27] Agreement on the Conservation of Albatrosses and Petrels. New Zealand's Spatially Explicit Fisheries Risk Assessment to determine if seabirds are at unacceptable risk gets reviewed. Retrieved from https://www.acap.aq/en/links/14-news/latest-news/2860-new-zealand-s-spatially-explicit-fisheries-risk-assessment-to-determine-if-seabirds-are-at-unacceptable-risk-gets-reviewed.

[28] Global Issues. Why is Biodiversity Important? Who cares? Retrieved from http://www.globalissues.org/article/170/why-is-biodiversity-important-who-cares.

implementation are to be National Biodiversity Strategies and Action Plans or equivalent instruments (NBSAPs).

There are five strategic goals and 20 ambitious yet achievable targets. Collectively known as the Aichi Targets, they are part of the Strategic Plan for Biodiversity. The five Strategic Goals are:

- Goal A – Address the underlying causes of biodiversity loss by mainstreaming biodiversity (NBSAPs) across government and society.
- Goal B – Reduce the direct pressures on biodiversity and promote sustainable use.
- Goal C – Improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity.
- Goal D – Enhance the benefits to all from biodiversity and ecosystem services.
- Goal E – Enhance implementation through participatory planning, knowledge management and capacity building.

Targets 6–11 specifically refer to fisheries and marine ecosystems and are provided in Section 18.6 of this chapter. New Zealand has responded and updated its Biodiversity Action Plan in 2016.[29]

18.2.2 GLOBAL MARINE ASSESSMENT[36]

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 September 2016.[30]

After many years of international negotiations to strengthen the science-policy interface on biodiversity and ecosystem services at all levels, more than 90 governments (including New Zealand) agreed in April 2012 to officially establish the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES).[31] IPBES is a leading global body providing scientifically sound and relevant information to support more informed decisions on how biodiversity and ecosystem services are conserved and used around the world. Currently a work programme is under construction to enable an integrated approach to analytics about biodiversity across the globe is under development.[32]

18.2.3 CLIMATE CHANGE MEANS OCEAN CHANGE

> '*Rapidly rising greenhouse gas concentrations are driving ocean systems toward conditions not seen for millions of years, with an associated risk of fundamental and irreversible ecological transformation. Changes in biological function in the ocean caused by anthropogenic climate change go far beyond death, extinctions and habitat loss: fundamental processes are being altered, community assemblages are being reorganized and ecological surprises are likely.*' (Quote from Global Issues. Climate change affects biodiversity. http://www.globalissues.org/article/172/climate-change-affects-biodiversity).

Public discussion about the impacts of climate change tend to focus on changes to land and the planet's surface or atmosphere. However, most of the excess heat is going into the oceans, and changes in the physical and chemical properties of seawater are underway (see Chapter 12).

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). A study of the global patterns of climate change impacts on ocean biodiversity projected the distributional ranges of a sample of 1066 exploited marine fish and invertebrates for 2050 using a newly developed dynamic

[29] DOC. New Zealand Biodiversity Action Plan 2016–2020. Retrieved from http://www.doc.govt.nz/nature/biodiversity/nz-biodiversity-strategy-and-action-plan/new-zealand-biodiversity-action-plan.

[30] United Nations World Ocean Assessment, http://www.worldoceanassessment.org.

[31] International Union for Conservation of Nature. Themes. Retrieved from http://www.iucn.org/what.

[32] IPBES. Work programme. Retrieved from http://www.ipbes.net/work-programme.

bioclimate envelope model. The study showed that climate change may lead to numerous local extinctions in the sub-polar regions, the tropics and semi-enclosed seas (Cheung et al. 2009). Simultaneously, species invasion is projected to be most intense in the Arctic and the Southern Ocean. With these elements taken together, the model predicted dramatic species turnovers of over 60% of the present biodiversity, implying ecological disturbances that potentially disrupt ecosystem services (Cheung et al. 2009).

The Intergovernmental Panel on Climate Change (IPCC) 5th Report (2014)[33] includes chapters to explicitly address ocean climate change issues for the first time. The Working Group I and Working Group II Contributions to the Fifth Assessment Report include chapters on the ocean (WG I) and Climate Change 2014: Impacts, Adaptation, and Vulnerability including Chapters on Coastal and Oceans ecosystems, and sections on biodiversity (WG II). Working Group I consider ocean biogeochemical changes, including ocean acidification in their Chapter 3 (Observations - Ocean), and in Chapter 6 on carbon and other biogeochemical cycles. Working Group II considered water property changes, including temperature and ocean acidification in Chapter 6, '*Ocean Systems*'. In addition, '*Carbon Cycle including Ocean Acidification*' were identified as cross-cutting themes across (predominantly) WG I and WG II.

The 5th IPCC report identifies that the effects of increasing greenhouse emissions – in particular carbon dioxide – on the oceans is likely to be significant. The basic chemistry of ocean acidification is well understood. These are the three main concepts:

1. More CO_2 in the atmosphere means more CO_2 in the ocean;
2. Atmospheric CO_2 is dissolved in the ocean, which becomes more acidic; and
3. The resulting changes in the chemistry of the ocean disrupts the ability of plants and animals in the sea to make shells and skeletons of calcium carbonate, while dissolving shells already formed.

The Summary for Policymakers (IPPC AR5, 2014) warns that ocean related climate change at the global scale overshadows the existing challenges of managing local impacts causing declines in marine biodiversity in the face of current levels of human use and impact. '*Due to projected climate change by the mid-21st century and beyond, global marine-species redistribution and marine-biodiversity reduction in sensitive regions will challenge the sustained provision of fisheries productivity and other ecosystem services (high confidence). Spatial shifts of marine species due to projected warming will cause high-latitude invasions and high local-extinction rates in the tropics and semi-enclosed seas (medium confidence)*'.

Hobday et al. (2006) reported on the relative risks and likely impacts of ocean climate change and ocean acidification to marine life in Australian waters. This approach was extremely useful for summarising risks and threats of climate change on marine systems to policymakers and the subsequent development of the Commonwealth Environment Research Facilities (CERF) Marine Biodiversity Hub in Australia (Figure 18.1).

The Hub analysed patterns and dynamics of marine biodiversity through four research programmes to determine the appropriate units and models for effectively predicting Australia's marine biodiversity. These programmes were designed to develop and deliver tools needed to manage Australia's marine biodiversity in a changing ocean climate. The final report from three years intense research is available at the website.[34] Australia also has the Marine Adaptation Network that comprises a framework of five connecting marine themes (integration; biodiversity and resources; communities; markets and policy) that cut across climate change risk, marine biodiversity and resources, socioeconomics, policy and governance, and includes ecosystems and species from the tropics to Australian Antarctic waters.[35]

In late June 2011, two science-based reports heightened concerns about the critical state of the world's oceans in response to ocean climate change. One focuses on the potential impacts of ocean acidification on fisheries and

[33] Greenhouse Gas Warming. IPCC 5th Report. Retrieved from http://www.global-greenhouse-warming.com/IPCC-5th-Report.html.

[34] Marine Biodiversity Hub, www.marinehub.org.

[35] NCCARF. About us. Retrieved from http://arnmbr.org/content/index.php/site/aboutus.

higher trophic level ecology and takes a modelling approach to scaling from physiology to ecology (Le Quesne & Pinnegar 2012) and the other assesses the critical state of the world's oceans in relation to climate change and other stressors (Rogers & Laffoley 2011). 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). They found that many of the biggest threats stemmed from human activities outside the marine environment itself. The two biggest threats 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.

Groups	Distribution/ Abundance	Phenology	Physiology/ Morphology/ Behaviour	Impacts on biological communities	Examples of impacts
Phytoplankton	High	High	Medium	High	Temperate phytoplankton province will shrink considerably
Zooplankton	High	High	Medium	High	Acidification will dissolve planktonic molluscs
Seagrasses	Medium	Low	High	Medium	Increased dissolved carbon dioxide may increase productivity
Mangroves	Medium	Low	Medium	High	Sea level rise will destroy mangrove habitat
Kelp	High	Medium	High	High	Ranges will shift southwards as SST warms
Rocky reefs	High	Medium	High	Low	Ranges will shift southwards as temperature warms
Coral reefs	High	Medium	High	High	Acidification and warming will cause calcification problems and coral bleaching
Cold water corals	High	Low	Low	High	Ocean acidification will dissolve reefs
Soft bottom dwelling fauna	Medium	Medium	Medium	Medium	Modified plankton communities or productivity will reduce benthic secondary production
Seafloor dwelling and demersal fishes	High	Medium	Medium	High	Southward movement of species along the east and west coast of Australia
Pelagic fishes	Medium	Low	Medium	Low	Pelagic tunas will move south with warming
Turtles	High	Medium	High	Low	Warming will skew turtle sex ratios
Seabirds	Medium	Medium	Low	Low	Shift in timing of peak breeding season as temperatures warm
Total number of high impact habitats or species groups	8	2	5	7	High impacts are expected for distribution, physiology and community processes

Figure 18.1: Potential biological impacts of climate change on Australian marine life. The ratings in this table are based on the expected responses to predicted changes in Sea Surface Temperature (SST), salinity, wind, pH, mixed layer depth and sea level, and from literature reviews for each species group. The implicit assumption underlying this table is that Australian marine species will respond in similar ways to their counterparts throughout the world (Hobday et al. 2006). Note: phenology means lifecycle. While generally accurate, new data are suggesting unanticipated or higher levels of threats for some taxa and habitats. Examples include that impacts of climate change on wind intensification on coastal upwelling systems is likely to be a strong driver of productivity, with implications for coastal and pelagic food webs (Sydeman et al. 2014), that geographic limits to species range shifts are predicted to limit the ability of species to shift in response to temperature change (Burrows et al. 2014), and that ocean acidification is receiving stronger attention in temperate coastal systems.

18.2.4 CENSUS OF MARINE LIFE 2000–10

In 2010, the international initiative to conduct a Census of Marine Life (CoML)[36] was concluded after 10 years of accessing and databasing existing records, sampling and exploration around the globe. The Census was an unprecedented collaboration among researchers from more than 80 nations to assess and explain the diversity, distribution, and abundance of life in the oceans. During the last decade, the 2700 scientists involved in the Census have mounted 540 expeditions, identified more than 6000 potentially new species, catalogued upward of 31 million distribution records, and generated 2600 scientific publications. NIWA scientists were part of the team that led CenSeam,[37] the seamount component of the Census of Marine Life, and scientists from NIWA and the University of Auckland played significant roles in a number of other programmes. The New Zealand International Polar Year-Census of Antarctic Marine Life (IPY-CAML) voyage to the Ross Sea in 2008 was a major contribution to CoML.

The Census facilitated activities that 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. Among the millions of specimens collected in both familiar and seldom-explored waters through new collecting, the Census found more than 6000 potentially new species and completed formal descriptions of more than 1200 of them. It also found that some species considered to be rare are commoner than previously thought (Ausubel et al. 2010). The digital archive (the Ocean Biogeographic Information System OBIS, http://www.iobis.org) has now grown to 31 million observations, and the Census compiled the first regional and global comparisons of marine species diversity. It helped to create the first comprehensive list of the known marine species, and also helped to compose web pages for more than 80 000 species in the Encyclopedia of Life.[38]

Applying genetic analysis on an unprecedented scale to a dataset of 35 000 species from widely differing major groupings of marine life, the Census graphed the proximity and distance of relations among distinct species, providing new insight into the genetic structure of marine diversity. With the genetic analysis often called barcoding, the Census sometimes decreased diversity but generally its analyses expanded the number of species, especially the number of different microbes, including bacteria and archaea.

The Census has overwhelmingly demonstrated that the total number of species in the ocean remain largely unknown. The Census also demonstrated that evidence of human impacts on the oceans extends to all depths and habitats and that we still have much to learn to integrate use of resources with stewardship of a healthy marine ecosystem. The Census results could logically extrapolate to at least a million kinds of eukaryotic marine life that earn the rank of species and to tens or even hundreds of millions of kinds of microbes. However, an expert assessment recently determined that only 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 summary of the overall state of knowledge about marine biodiversity after the Census by Costello et al. (2010) places New Zealand sixth out of 18 national regions based on the collective knowledge assembled by the Census National and Regional Implementation Committees (NRIC) and comparing 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 (Figure 18.2).

All NRICs reported what they considered the main threats to marine biodiversity in their region, citing published data and expert opinions. Although the reports were not standardised, the threats identified were grouped into several overarching issues. The data on biodiversity threats were integrated so as to rank each threat from 1 (very low) to 5 (very high threat) in each region. New Zealand was

[36] Census of Marine Life. Results and Publications. Retrieved from www.coml.org/results-publications.

[37] Census of Marine Life. Global Census of Marine Life on Seamounts (CenSeam). Retrieved from www.coml.org/global-census-marine-life-seamounts-censeam.

[38] Encyclopedia of Life, www.eol.org.

placed 12th out of 18 regions in terms of overall threat levels to biodiversity, overfishing and alien species invasion. 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).

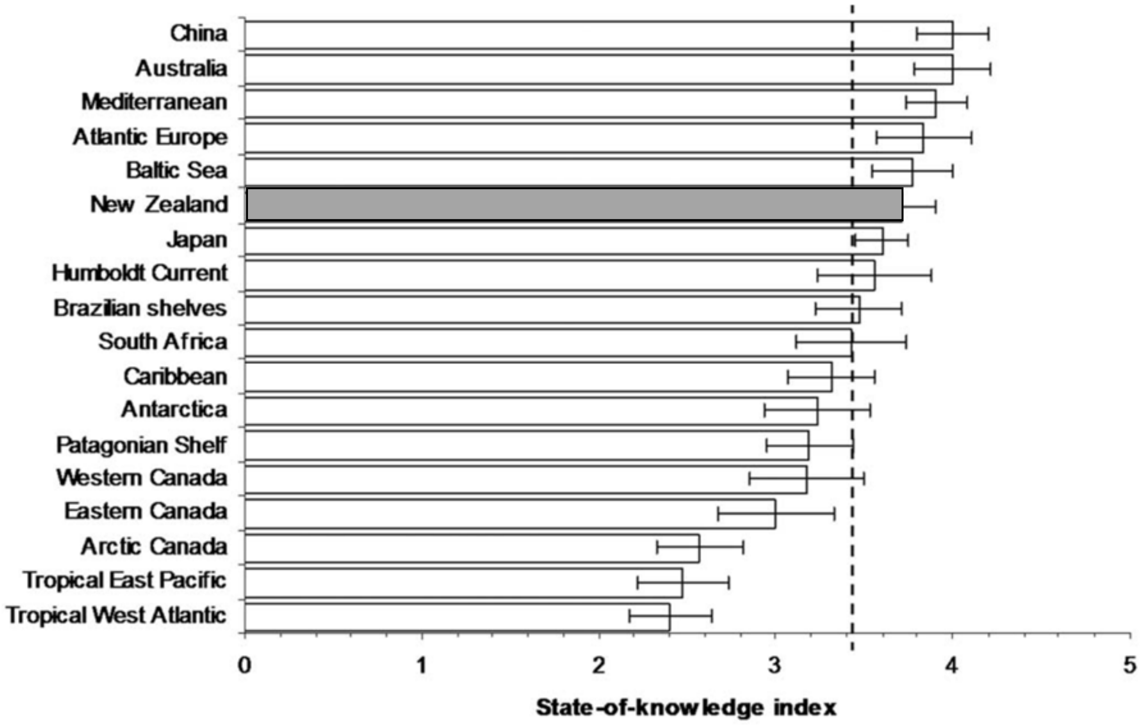


Figure 18.2: (from Costello et al. 2010). The regions are ranked by their state-of-knowledge index (mean ± standard error) across taxa. Dashed line represents the overall mean.

18.2.5 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. Marine biodiversity indicators have been developed for the UK and the EU.[39] Marine environmental monitoring networks have been developed in the USA, Canada, Australia and South Africa. Global networks include the Global Ocean Observing System (GOOS), which is a permanent global system for observations, modelling and analysis of marine and ocean variables; Global Climate Observing System (GCOS[40]), which stimulates, encourages, coordinates and otherwise facilitates observations by national or international organisations. A Southern Ocean Observing System (SOOS[41]) is up and running. The SOOS International Project Office was officially opened in August 2011, hosted by the Institute for Marine and Antarctic Studies (IMAS) at the University of Tasmania, Australia.[42]

[39] Joint Nature Conservation Committee. The UK Biodiversity Indicators. Retrieved from http://jncc.defra.gov.uk/page-4233.

[40] The Global Ocean Observing System. GOOS Projects. Retrieved from www.ioc-goos.org/index.php?option=com_content&view=article&id=12&Itemid=26&lang=en.

[41] Southern Ocean Observing System, http://www.soos.aq.

[42] Scientific Committee on Antarctic Research. Southern Ocean Observing System. Retrieved from http://www.scar.org/soos.

Others include:

- ARGO, an international deepwater monitoring system of free floating buoys that are part of the integrated global observation strategy and that New Zealand makes a significant contribution to in the Pacific Ocean.[43]
- The Continuous Plankton Recorder (CPR) Surveys have been collecting data from the North Atlantic and the North Sea on the ecology and biogeography of plankton since 1931.[44] Sister CPR surveys around the globe include the SCAR SO-CPR Survey established in 1991 by the Australian Antarctic Division to map the spatial-temporal patterns of zooplankton and then to use the sensitivity of plankton to environmental change as early warning indicators of the health of the Southern Ocean. It also serves as reference for other monitoring programs such as CCAMLR's Ecosystem Monitoring Program C-EMP and the developing Southern Ocean Observing System. New Zealand contributes to the Southern Ocean CPR data collection through the BRAG.
- New Zealand has now formed a partnership with Australia's Integrated Marine Observing System (IMOS[45]), which was established in 2007. IMOS is designed to be a fully integrated national array of observing equipment to monitor the open oceans and coastal marine environment around Australasia, covering physical, chemical and biological variables. All IMOS data is freely and openly available through the IMOS Ocean Portal for the benefit of Australian and New Zealand marine and climate science as a whole.
- Oceans 2025 [46] is an initiative of the Natural Environment Research Council (NERC) funded Marine Research Centres. This addresses environmental issues that require sustained long-term observations.
- The Global Ocean Acidification Observing Network (GOA-ON) is an existing global ocean carbon observatory network of repeat hydrographic surveys, time-series stations, floats and glider observations, and volunteer observing ships. The interactive map below offers the best information available on the current inventory of global OA observing platforms. With participation from scientists from over 30 countries, GOA-ON is a strong foundation of observations of the carbonate chemistry targeted to understand chemical and ecological changes resulting from ocean acidification from regions throughout the world.

New Zealand has now developed the NZ-AON (NZ Ocean Acidification monitoring Network), which was initiated in 2014 and now has 11 sampling sites around NZ. This is run by NIWA & the University of Otago. In 2017 New Zealand government has also joined the Ocean Acidification Alliance.[47]

A challenge for MPI, other government agencies and New Zealand is how to assimilate any or all of the above monitoring approaches to assess the nature and extent of biodiversity changes, and to assess the effectiveness of management measures to protect or enhance biodiversity or halt its decline.

18.2.6 VALUATION OF BIODIVERSTIY AND ECOSYSTEMS

The national and global responsibility for New Zealand to maintain a strong environmental record in fisheries and other marine-based industries is increasing. There is growing awareness of international treaties and agreements that New Zealand is party to. Global markets are becoming increasingly sensitive to our national environmental record. Fishing companies who meet rigorous standards receive Marine Stewardship Council Certification for certain fisheries (currently, hoki trawl,

[43] Fisheries and Oceans Canada. Preliminary Assessment of Oceans Observing Systems Value. Retrieved from http://www.qc.dfo-mpo.gc.ca/publications/science/evaluation-assessment-eng.asp.
[44] Sir Alister Hardy Foundation for Ocean Science, https://www.sahfos.ac.uk.
[45] Integrated Marine Observing System, http://imos.org.au.
[46] Oceans 2025, http://www.oceans2025.org.
[47] International Alliance to Combat Ocean Acidification, https://www.oaalliance.org/current-members.

southern blue whiting pelagic trawl and albacore tuna troll fisheries). Proposals to exploit other living marine resources or extract non-living marine resources are increasingly under scrutiny to ensure that such activities do not adversely degrade the marine environment or impact on marine living resource industries such as fishing and aquaculture.

The invisibility of biodiversity values has often encouraged inefficient use or even destruction of the natural capital that is the foundation of our economies. A recent international initiative 'The Economics of Ecosystems and Biodiversity' (TEEB) [48] demonstrates the application of economic thinking to the use of biodiversity and ecosystem services. This can help clarify why prosperity and poverty reduction depend on maintaining the flow of benefits from ecosystems; and why successful environmental protection needs to be grounded in sound economics, including explicit recognition, efficient allocation, and fair distribution of the costs and benefits of conservation and sustainable use of natural resources. Valuation is seen as a tool to help recalibrate the economic compass that has led to decisions about the environment (and biodiversity) that are prejudicial to both current well-being and that of future generations.

The idea of putting monetary values on natural capital is not without controversy; many underpinning life-supporting services such as nutrient recycling are difficult to quantify at scale, and assigning dollar values to many types of ecosystem services is fraught with difficulty.

A new United Nations platform, the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services), 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. This new international platform is similar in function to the 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 past 750 years of human activity have 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 near population centres, as well as in more remote areas and to depths in excess of 1000 m (MacDiarmid et al. 2014a, Ministry for Primary Industries 2012). In some cases by looking back over historical records it becomes apparent how much biodiversity loss has occurred. Over long time spans incremental impacts can lead to major shifts in biodiversity composition. An analysis of marine biodiversity decline over a couple of decades could miss the major changes that can occur incrementally over long periods.

While New Zealand has reasonable archaeological, historical and contemporary data on the decline in abundance of individual marine species, in some cases over a period of 750 years, 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.

A re-evaluation of the threat status of New Zealand's marine invertebrates was undertaken by the Department of Conservation in 2009, and identified no taxa that had improved in threat status as a result of past or ongoing conservation management action, nor any taxa that had worsened in threat status because of known changes in their distribution, abundance or rate of population decline (Freeman et al. 2010). The authors cautioned however that only a small fraction of New Zealand's marine invertebrate fauna had been evaluated for their threat status and that many taxa remain 'data deficient' or unlisted. In June 2013, the Department of Conservation held an expert workshop to assess New Zealand's marine invertebrates using the

[48] TEEB (2010) The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB. Retrieved from www.teebweb.org.

New Zealand Threat Classification System (NZTCS) criteria, updating a previous listing process from 2009 (Freeman et al. 2010). A number of changes were made to the list, including those resulting from taxonomic name changes (Freeman et al. 2013). A total of 415 taxa were assessed, with a number of changes made to threat categories to reflect changes in certainty or knowledge around the distribution, abundance and population trends of some taxa, or to reflect a reinterpretation of the available data. All marine invertebrates assessed in 2009 were reassessed and an additional 108 taxa were also assessed. Most of the latter were assessed as either data deficient, or naturally uncommon. One taxon that was included in the list produced in 2009 was excluded from the current listing – *Cellana strigilis bollonsi* Powell, 1955, which is now considered to be a synonym of *C. oliveri*. While a number of changes were made to the threat categories assigned to the marine invertebrates we assessed, just one change was the result of an actual decline in abundance. The brachiopod *Pumilus antiquatus* (a monotypic, endemic genus) was listed as Nationally Endangered in 2009, but listed as Nationally Critical in 2013, to reflect an apparent decline in abundance at the sites it has previously been recorded from (Otago Harbour and Lyttelton). No taxa improved in status between 2009 and 2013 as a result of an actual change in distribution or abundance. Six taxa were listed as Nationally Critical and an additional five taxa were also included in the Threatened category. The majority of taxa we assessed were classified as At Risk, with most of these being taxa that are naturally uncommon, such as island endemics. A large number of marine invertebrates were assessed as Data Deficient. However, the majority of the New Zealand marine invertebrate fauna (over 95%) remains unassessed in the New Zealand Threat Classification System. While representatives of phyla not assessed in 2009 were included in the current assessment (e.g., sponges, phylum porifera), the full range of marine invertebrate phyla are not yet represented in the list.

A re-evaluation of marine mammal threat status found that relative to the previous listing, the threat status of two species worsened: the NZ sea lion (*Phocarctos hookeri*) was uplisted to Nationally Critical and the bottlenose dolphin (*Tursiops truncatus*) was uplisted to Nationally Endangered. No species was considered to have an improved status (see chapters on marine mammals; Baker et al. 2010).

New Zealand has 92 seabird and 14 shorebird species and subspecies (taxa) – the highest number of endemic seabirds (found only in a particular area) in the world. Nearly 25% of the world's seabird species breed in the New Zealand region, and almost 10% breed only here. Seabirds and shorebirds tend to be at or near the top of the food chain, and thrive only if the marine ecosystem is healthy. Decreasing bird populations can signal the ecosystem is degrading.

In 2012, DOC classified Conservation status of seabirds and shorebirds as a case study (see definition: http://archive.stats.govt.nz/browse_for_stats/environment/environmental-reporting-series/environmental-indicators/Home/About.aspx#topics).

The key findings were that about one-third (32 of 92) of New Zealand's resident indigenous seabird taxa and more than half (8 of 14) of our resident indigenous shorebird taxa are threatened with extinction. In addition:

- more than half of our resident indigenous seabird taxa (51 of 92) are at risk of extinction
- more than one-quarter (29%) of our resident indigenous shorebird taxa (4 of 14) are at risk of extinction.

Further information is presented in MfE's Environmental report on the marine environment released in October 2016 (see http://www.mfe.govt.nz/publications/marine-environmental-reporting/our-marine-environment-2016%20).

Key marine environment and biodiversity related issues are outlined by MfE in Environment Aotearoa, at http://www.mfe.govt.nz/publications/environmental-reporting/environment-aotearoa-2015.

The most recent summary of knowledge about marine biodiversity in New Zealand is provided by Gordon (2009, 2010, 2012) and Gordon et al. (2010). Table 18.1 gives a tally of 17 987 living species in the EEZ, including 4320 known undescribed species in collections. More recent updates using all records available within OBIS, NIWA and Te Papa collections 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 (Table 18.1).

In New Zealand, a number of new marine research projects were initiated in 2012 that 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' (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).

Other MBIE-funded projects with marine components include Climate Change Impacts and Implications, which was completed in 2016.[49] The key results from the marine case study are summarised in a synthesis report.[50] Another significant programme of work funded by MBIE that MPI has contributed to includes 'Coastal Acidification: Rate, Impacts & Management'.[51]

One of the largest marine research developments in 2014 was the launch of the National Science Challenge 'Sustainable Seas'.[52] The Challenge aims to enhance the utilisation of our [NZ] marine resources within environmental and biological constraints.

To do this, the Sustainable Seas Challenge will:

- Assess the cultural, economic and environmental values of our oceans and coasts
- Investigate and describe the impacts of natural and human stresses on marine ecosystems
- Identify options for environmental mitigation or restoration; and
- Overcome impediments to enhanced resource use.

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.

At the time of writing, NIWA is reviewing its work programme for the Fisheries Centre, Aquaculture Centre, Te Kuwaha and the Coasts and Oceans Centre under the newly launched Marine Platform that replaces MBIE core funding. Some of the projects under these work programmes dovetail closely with Sustainable Seas and are considered to be aligned to the Challenge.

[49] Climate Changes, Impacts and Implications for New Zealand. Impacts on the ocean food chain: implications for native biodiversity, seafood & tourism industries. Retrieved from http://ccii.org.nz/research-aims/ra2/marine.

[50] Climate Changes, Impacts and Implications for New Zealand to 2100 (2016) Synthesis Report: RA2 Marine Case Study. Retrieved from http://ccii.org.nz/wp-content/uploads/2017/04/RA2-MarineCaseStudySynthesisReport.pdf.

[51] NIWA. CARIM (Coastal Acidification: Rate, Impacts & Management). Retrieved from https://www.niwa.co.nz/coasts-and-oceans/research-projects/carim-coastal-acidification-rate-impacts-management.

[52] Beehive. Sustainable Seas National Science Challenge launched. 4 September 2014. Retrieved from http://www.beehive.govt.nz/release/sustainable-seas-national-science-challenge-launched.

18.3 THE MPI BIODIVERSITY RESEARCH PROGRAMME

The recognition of increasing societal expectation to use fisheries management measures that will achieve biodiversity conservation was signaled by MPI through Fisheries 2030 [53] 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'. While New Zealand's environmental performance with regard to fishing is perceived to be relatively high on an international scale, MPI is not complacent about the ongoing requirement to monitor and provide evidence that measures to achieve biodiversity conservation needs are being met. This is particularly true of the need to better understand and mitigate the effects of fishing in the areas impacted by fishing. The effects of fishing on the aquatic environment and risks to biodiversity and the aquatic environment are supported through MPI's Deepwater Research Plan, as well as the Aquatic Environment and Biodiversity Research Programmes.

There are also 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 (see footnote 10). These include aesthetic and cultural values as well as other economic values such as tourism and marine recreation other than fishing.[54] To link socioeconomic values of biodiversity to science supporting fisheries management will require a multi-disciplinary approach only just beginning in New Zealand.

MPI responded to the NZBS in 2000 with the establishment of the MPI Biodiversity Programme, which has run successfully for more than 10 years with 64 research projects and a large number of published outputs, presentations and contributions to New Zealand and CCAMLR management measures.

The Ministry is also one of several New Zealand 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. MPI 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. Research conducted under the BioRoss component of the MPI Biodiversity Programme seeks to help New Zealand deliver on its international obligations to support an ecosystem-based approach to management in Antarctic waters. There are strong links with the MPI Antarctic Working Group research and with other Ross Sea ecosystems research carried out by NIWA Fisheries, and Coast and Oceans Centres under the Marine Platform (e.g., Sharp et al. 2010).

[53] Ministry of Fisheries. Fisheries 2030. Retrieved from https://fs.fish.govt.nz/NR/rdonlyres/40170C41-1B75-4765-A1D9-B4DB250BA53E/0/fisheries2030report.pdf.

[54] MARBEF: The Valencia Declaration 2008, http://www.marbef.org/worldconference.

Table 18.1: Diversity of marine species found in the New Zealand region (after Gordon 2010, 2012, Gordon et al. 2010 and current unpublished NIWA data). [Continued on next page]

Taxonomic group	No. species[1]	State of knowledge (1 low, 5 high)	No. alien species naturalised	No. experts	No. ID guides[2]
Superkingdom Prokaryota	82	1–2	>1	3	1
Cyanobacteria	40	3–4	1	2	1
All other Bacteria	42	1	?	1	0
Superkingdom Eukaryota	17 905	3–4	185	59	77
Kingdom Protozoa	53	2	4	5	4
Kingdom Chromista	2 541	3–4	14	7	3
Ochrophyta	858	3–4	11	1	2
Miozoa (incl. dinoflagellates)	249	3–4	0	2	0
Retaria (incl. foraminifera)	1 217	4–5	3	2	3
All other Chromista	217	2–3	0	1	0
Kingdom Plantae	711	4–5	15	7	6
Chlorophyta	156	3–4	0	2	2
Rhodophyta	550	4	12	2	2
Tracheophyta	5	5	3	3	2
Kingdom Fungi	89	3	1	2	0
Kingdom Animalia	14 511	3–4	150	40	68
Porifera	770	3	7	1	5
Cnidaria	1 114	4	24	0	7
Platyhelminthes	324	2	2	1	3
Mollusca	3 595	4	15	4	2
Annelida	793	3–4	33	1	2
Bryozoa	957	3–4	29	2	4
Arthropoda (esp. Crustacea)	2 979	4	27	11	17
Echinodermata	636	5	0	3	6
Tunicata	193	4	3	1	6
All other invertebrates	1 723	2–5	4	5	12
Fishes	1 254	4–5	6	6	8
All other vertebrates	173	5	0	4	4
TOTAL REGIONAL DIVERSITY[3]	17 987	3–4	186	62	78

[1] Sources of the tallies: scientific literature, books, field guides, technical reports, museum collections.
[2] Identification guides cited in Gordon et al. (2010).
[3] Totals from Gordon (2009, 2010, 2012, 2013), Gordon et al. (2010) and Nelson (2014) and unpublished NIWA data.

The Biodiversity Research Programme is guided by a multi-stakeholder biodiversity research advisory group (BRAG), chaired by MPI. The research commissioned for the period 2001–05 reflected goals set by the NZBS and the BRAG, while remaining compatible with the Ministry of Fisheries Statements of Intent (SOIs). During the first three years of this period, the Ministry of Fisheries also commissioned marine biosecurity research under NZBS, but this was transferred to Biosecurity New Zealand (MAFBNZ) in 2004. From 2006 to 2010, the programme evolved further with the development of a five-year work programme to address shortcomings identified in a review of the NZBS by Green and Clarkson (2006). An overview of the current Biodiversity Programme at a glance is given in Figure 18.3.

18.3.1 OVERALL PROGRESS IN MPI MARINE BIODIVERSITY RESEARCH

The MPI Biodiversity Research Programme has three overarching science goals:

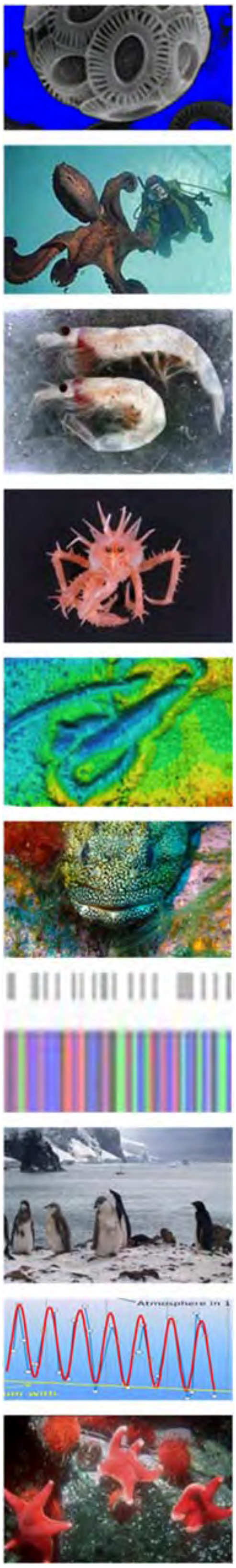
- To describe and characterise the distribution and abundance of fauna and flora, as expressed through measures of biodiversity, and improving understanding about the drivers of the spatial and temporal patterns observed.

- To 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.
- To identify which components of biodiversity are required to ensure the sustainability of healthy marine ecosystems as well as to meet societal values on biodiversity.

More specific Science Objectives developed below have been modified by BRAG over time and are used to focus the research commissioned:

1. To classify and characterise the biodiversity, including the description and documentation of biota, associated with near-shore and offshore marine habitats in New Zealand.
2. To develop ecosystem-scale understanding of biodiversity in the New Zealand marine environment.
3. To investigate the role of biodiversity in the functional ecology of near-shore and offshore marine communities.
4. To assess developments in all aspects of diversity, including genetic marine biodiversity and identify key topics for research.
5. To determine the effects of climate change and increased ocean acidification on marine biodiversity, as well as effects of incursions of non-indigenous species, and other threats and impacts.
6. To develop appropriate diversity metrics and other indicators of biodiversity that can be used to monitor change.
7. To identify threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation.

To date, 64 research projects have been commissioned. Early studies focused primarily on Objectives 1 and 2 and resulted in reviews, Identification Guides, habitat and community characterisations, and revised taxonomy for certain groups of organisms. These objectives have also resulted in 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. Over time, the complexity and scale of studies has increased with projects on the functional ecology of marine ecosystems from localised experimental manipulation to broad-scale observations across hundreds of square kilometres under Objective 3. Such studies have also pursued the development of improved measures of biodiversity and indicators under Objectives 6 and 7. A study on changes in shelf ecosystems over the past 1000 years is yielding insights into the effects of long-term climate change, land-use effects and fishing on marine ecosystems while more recently, some studies have begun to address the effects of ocean acidification on marine biodiversity under Objective 5. A study underway has reviewed genetic variation in the New Zealand marine environment and is conducting field observations on several species to examine genetic variation across latitudinal gradients. Aspects of the seven Objectives have also been addressed through a range of biodiversity projects in the Ross Sea region including the International Polar Year Census of Antarctic Marine Life project (IPY-CAML). A key to study findings is consideration of biodiversity within the context of the carrying capacity of the system and the natural assemblages of biota supported by that system in the absence of human disturbance.



BIODIVERSITY THEMES	KEY QUESTIONS
BIODIVERSITY PATTERNS & DISTRIBUTION • Fauna and flora (taxonomy, biosystematics) • Distribution & abundance of major groups • Reviews of existing knowledge • Biogeography • Drivers of observed patterns	• What is the abundance and distribution of marine biodiversity in NZ? • What are the key drivers of observed patterns in biodiversity? • How much marine endemism is there in NZ waters? • What is the organism size distribution? • How do patterns in biodiversity change over time? • What is the state of marine biodiversity in NZ?
HABITAT DIVERSITY • Biogenic reefs • Rocky reefs • Rhodolith beds • Seamounts • Soft sediments • Habitat mapping EEZ • Deepsea habitats • Physical and biological characterisation	• What are the relative goods and services offered by each habitat to aquatic environment health? • Can the assemblages and biodiversity of marine habitats in the EEZ be predicted by modelling? • Which habitats are at greatest risk from extraction practices? • What proportion of a given habitat needs to remain intact for healthy ecosystem functioning?
FUNCTIONAL DIVERSITY • The role of different animal/plant groups in the ecosystem • Trophic processes • Bentho-pelagic processes	• How does biodiversity contribute to the resilience of ecosystems to perturbation? • Can we use ecosystem function to classify biodiversity? • Which key processes need to be retained?
GENETIC DIVERSITY • Barcode of Life • Connectivity (populations, areas)	• What are the barriers to connectivity within species? • What is the role of endemism in characterising the evolutionary history and taxonomy?
THREATS TO BIODIVERSITY • Climate change and variability • Invasive organisms; fishing • Land-use effects • Cumulative effects	• What are the key threats? • Does biodiversity increase resilience to climate change? • Which components of the ecosystem will be most at risk from climate change? • Will climate change offer any opportunities to the seafood sector?
METHODS • Measuring biodiversity • Classification • Predictive modelling • Biodiversity indicators • Monitoring biodiversity • Ecosystem approaches	• How can we best measure and portray biodiversity? • How scalable are results from a local scale to an ecosystem scale? • What do we need to monitor to measure risks and change to ecosystem health? • How can we measure the economic value of biodiversity and ecosystem services?
BIOROSS/ & IPY RESEARCH • Bioross coastal biodiversity • Subtidal ice-sea interface • Census of Antarctic marine Life survey for IPY, Ross Sea • Trophic modelling Ross Sea • Balleny Islands survey for MPA • Functional habitats	• What is the connectivity between biodiversity in the Ross Sea and NZ? • How are biota adapted to polar conditions and what is their sensitivity to perturbation? • Are MPAs a useful protection tool for the Ross Sea? • Are climate change effects on the ocean already impacting on the Ross Sea biota?

Figure 18.3: Summary of MPI Biodiversity Research Programme 2000–16. [Continued on next page]

ACHIEVEMENTS & KNOWLEDGE TO DATE	CURRENT WORK
• Taxonomy of coralline algae and bryozoans (2 ID Guides) • Coral and seapen ID Guides produced • Review of macroalgae distribution on soft sediments • Contribution to several books on marine biodiversity in NZ • EEZ surveys on Fjordland, Spirit's Bay, Kermadec seamounts, Farewell Spit, Norfolk Ridge, Chatham Rise and Challenger Plateau. • Links to MPI biosecurity mapping; MEC, BOMEC • Extensive new data sets and specimen collections obtained • Sample lots from BPAs characterised	• Ongoing taxonomic work in relation to deep sea corals (VMEs) • Ongoing taxonomic work on specimens collected throughout the EEZ. • New survey planned for benthic biodiversity on the Chatham Rise
• Ecological input to improve MEC (fish, benthic invertebrates) • Deep-sea habitats , biogenic habitat and soft-sediment reviewed • Ocean Survey 20/20 habitats mapped Chatham-Challenger • Biodiversity of Kermadec and Chatham Rise seamounts mapped • Foveaux Strait habitats mapped • Classification of seamounts and VMEs developed • Testing of MEC with Chatham Challenger data • Rhodolith beds as havens of biodiversity in NZ	• Mapping biogenic structures • Mapping deepsea fisheries habitats in relation to ocean acidification threats from changing saturation horizons • Modelling benthic impacts • Macroalgae as indicators of ecosystem health • Southern coralline algae
• Rocky reef ecosystem function studied • Chatham Rise fish feeding study completed • Productivity in horse mussel/echinoderm benthic communities known • Bioindicators in estuarine systems in Otago determined • Chatham-Challenger functional component analysis completed • Shellhash habitat function in the coastal zone • Ocean acidification effects on shellfish, deep sea corals studied	• Sub lethal effects of OA on fish behaviour underway • Microcosm experiments on the effects of temperature and OA on productivity • Review of climate change threats and opportunities
• Molecular ID of certain fish and plankton determined • EEZ and Ross Sea species added to Barcode of Life Database. • Genetic assessment of ocean microbe diversity • Seamount connectivity reviewed	• Connectivity among coastal shellfish fish populations • Connectivity of VME fauna in EEZ waters and to the east
• Threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation identified • Monitoring of plankton on transect NZ to Ross Sea annually 5yrs • Changes in coccolithophore diversity and abundance in NZ waters and predicted change as temp and acidification increase assessed • Long-term effects of climate change on shelf ecosystems determined	• Experimental response of shellfish pH and temp. • CPR monitoring (plankton) • Acidification in deepwater fish habitat • Microcosm experiments OA
• Diversity metrics and other indicators to monitor change developed • Large-scale sampling protocols for habitat mapping determined • Acoustic habitat mapping tools developed • Workshop on qualitative modelling in marine environment Development of "OFOP" and DTIS-visual analytical methods • Predictive modelling progressed for biodiversity on different scales • Development of data to end-user portal interfaced with NABIS • Development of functional biota model for habitat classification • Tier 1 stat development for the Ocean change and Marine biodiversity	• Quantifying predictive modelling VMEs and benthic habitats • Benthic biodiversity biomass pre and post earthquake in Kaikoura Canyon • Quantifying LEK • Developing isoscape horizons
• Latitudinal gradient project and ICECUBE completed in Ross sea • Fish taxonomy and ID guide developed for the Ross Sea • Foodweb and role of silverfish vs krill studied • IPY-CAML 2008; Ross Sea 2006, BioRoss 2004 surveys done • Subtidal and offshore biodiversity sampled, Balleny Islands 2006 • Seaweed diversity determined at Balleny Islands • Bioregionalisation of the Ross Sea region completed • Review of squids, octopus MSc complete	• Uptake of biodiversity results to CCAMLR trophic modelling and biomass estimation, VMEs • Humpback whale project (part of Whale consortium) explores connectivity between NZ and Ross Sea

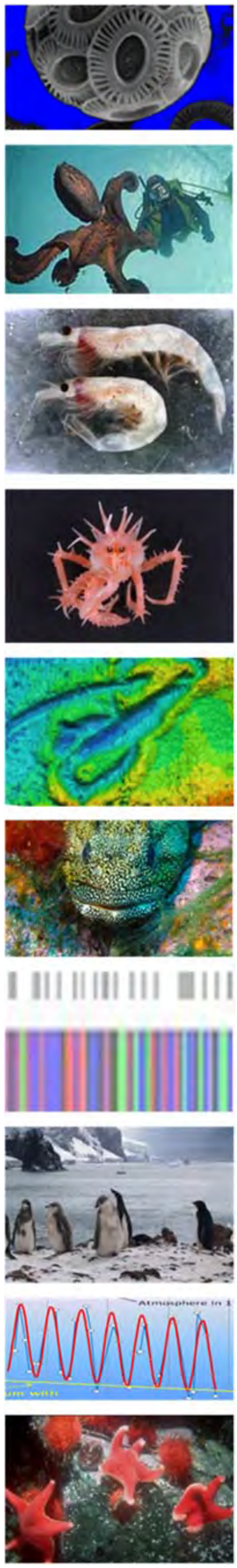


Figure 18.3 [Continued]: Summary of MPI Biodiversity Research Programme 2000–16.

Progress in the BRAG research programme increasing conceptual complexity from cataloguing of biodiversity to increasingly complex understanding of environmental drivers and functionality of biodiversity; and ultimately methods to develop standards and protection of biodiversity. A full list of projects can be obtained from Appendix 19.9 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.

Biodiversity research based in Antarctica lags behind EEZ-based research, simply because of the difficulty in securing additional funding to access and work in such a remote and hostile marine environment.

18.3.2 PROGRESS ON SCIENCE OBJECTIVE 1. 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), literature reviews, taxonomic studies and habitat mapping surveys have been undertaken.

REVIEWS AND BOOKS

The following lists scientific reviews and books on biodiversity that were commissioned by the programme:

ZBD2000-01 A review of current knowledge describing the biodiversity of the Ross Sea region (Bradford-Grieve & Fenwick 2001, 2002, Fenwick & Bradford-Grieve 2002a, 2002b, Varian 2005).

ZBD2000-06 The Living Reef: The Ecology of New Zealand's Rocky Reefs (eds. Andrew & Francis 2003).

ZBD2000-08 A review of current knowledge describing New Zealand's Deepwater Benthic Biodiversity (Key 2002).

ZBD2000-09 Antarctic fish taxonomy (Roberts & Stewart 2001).

ZBD2001-02 Documentation of New Zealand Seaweed (Nelson et al. 2002).

ZBD2001-04 Deep New Zealand (Batson 2003)

ZBD2001-05 Crustose coralline algae of New Zealand (Harvey et al. 2005, Farr et al. 2009, Broom et al. 2008)

ZBD2001-06 Biodiversity of New Zealand's soft-sediment communities (Rowden et al. 2012b).

ZBD2003-09 Macquarie Ridge Complex Research Review (Grayling 2004).

ZBD2008-27 Scoping investigation into New Zealand abyss and trench biodiversity (Lörz et al. 2012a).

In addition a major work that includes marine species – 'The New Zealand Inventory of Biodiversity' (Gordon 2009, Gordon 2010, Gordon 2012), has been completed. Field identification guides have also been published by MPI on deepsea invertebrates (projects ENV2005-20 and ZBD2010-39, Tracey et al. 2005, 2007, 2011b), bryozoans (project IPA2009/14, Smith & Gordon 2011) and on fish species (IDG2006-01, McMillan et al. (2011a, 2011b, 2011c), which further contribute to the accurate monitoring and identification of biodiversity in New Zealand waters.

PROJECTS

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); MSI's Seamount Programme, mainly commissioned through public-good science, supplemented by MPI projects ZBD2000-04 (e.g., Rowden et al. 2002, 2003), ZBD2001-10 (Rowden et al. 2004), ZBD2004-01 (Rowden & Clark. 2010), and MPI 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); soft sediment environments ZBD2003-08 (Neill et al. 2012); rhodolith community study ZBD2009-03 (Nelson et al. 2012); offshore surveys of the Chatham Rise and Challenger Plateau funded through Ocean Survey 20/20 programme, ZBD2006-04 (Nodder 2008) and ZBD2007-01 (Nodder et al.

2011, Hewitt et al. 2011a, 2011b, Bowden 2011, Bowden & Hewitt 2012, Bowden et al. 2011a, 2011b, 2011c).

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, 2006a), ZBD2003-03 (Rowden et al. 2012a, 2013a), ZBD2005-03 (MacDiarmid & Stewart 2012), ZBD2006-03 (Cummings et al. 2003, 2006b;), ZBD2008-23 (Nelson et al. 2010) and IPY2007-01 (Bowden et al. 2011a, Clark et al. 2010b, Eakin et al. 2009, Hanchet, et al. 2008a, 2008b, 2008c, 2008d, Hanchet 2009, 2010, Koubbi et al. 2011, Lörz et al. 2009, Mitchell 2008, O'Driscoll 2009, O'Driscoll et al. 2011, 2012, O'Loughlin et al. 2011).

HABITAT DIVERSITY, CLASSIFICATION AND CHARACTERISATION

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 MPI 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 2006, three projects to map coastal biodiversity were completed in the Coromandel scallop, Foveaux Strait oyster and southern blue whiting fisheries 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. MPI project (BEN2006-01) aimed to further optimise the MEC by producing a methodology for a Benthic Optimised MEC (Leathwick et al. 2010). MPI Ecological studies to improve habitat classification and vulnerability indices have also been completed through MPI AEWG projects on seamounts (ENV2005-15, ENV2005-16) (e.g., Clark et al. 2010c), and to supplement other studies funded by MPI 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 (Leathwick et al. 2010); macroalgal diversity associated with soft sediment habitats ZBD2008-05; deepsea benthic biodiversity in trench, canyon and abyssal habitats below 1500 m depth ZBD2008-27; distribution and associated biodiversity ZBD2009-03; and Chatham-Challenger project ZBD2007-01 (Hewitt et al. 2011a, 2011b, Bowden et al. 2011b, Compton et al. 2012).

Progress advanced considerably in recent years with the introduction of the whole-of-government Ocean Survey 20/20 Programme and Biosecurity New Zealand mapping projects (Beaumont et al. 2008, 2010). In addition, MPI implemented spatial management tools (Benthic Protection Areas[55]), on the basis of the Marine Environment Classification[56] to address broader statutory responsibilities on the environmental effects of fishing on biodiversity.

Current and recently started projects are as follows:

ZBD2012-03: Chatham Rise Benthos Ocean Survey 20/20

This project has two objectives: (1) to determine whether there are quantifiable effects on the benthos of gradients of fishing intensity, and (2) to document seabed habitats and fauna in previously unsampled areas of the central crest of Chatham Rise, particularly within the central Chatham Rise Benthic Protection Area (BPA). A single research voyage was undertaken in June 2013 (TAN1306) funded by OS 20/20, MPI, and NIWA, with additional funding from Chatham Rock Phosphate Ltd. (CRP) for research on the crest of the rise (Objective 2). For Objective 1, three towed-camera transects and three multicorer samples were collected from each of six 10 x 10 km survey 'boxes' south of Mernoo Bank and five survey boxes on the

[55] MPI. Benthic protection areas. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/protected-areas/benthic-protection-areas.

[56] MfE. Marine Environmental Classification (2005). http://www.mfe.govt.nz/more/environmental-reporting/reporting-act/marine/classification-systems.

southern flank of Chatham Rise along gradients of fishing intensity. For Objective 2, three camera transects were run in each of eight survey boxes within the Central Chatham Rise Benthic Protection Area (BPA).

Objective 1: Effects on the benthos of gradients of bottom-contact fishing intensity.

This component of the research is still in progress, with all datasets planned to be ready for statistical analyses to start in early 2015. To date, 1202 individual seabed photographs have been analysed to extract quantitative measures of benthic epifauna, bioturbation marks (burrows, tracks, etc.), and substratum type. Continuous video imagery is currently being analysed to capture quantitative data on larger, more sparsely occurring benthic epifauna and bioturbation, and benthic and demersal fishes. Multicorer sediment samples have been analysed for sediment grain size and chemistry, and macro-infauna have been sorted from the sediments and are being identified and counted by taxonomists. Meiofauna samples were also collected from the sediment cores and these are being analysed by the University of Otago.

When complete, the multi-scale data from these analyses, ranging from cm^2 (meio- and macro-infauna, and sediments from multicorer samples), to m^2 (epifauna and substrata from still imagery), 100s m^2 (epifauna, substrata and fishes from video imagery), and km^2 (multibeam echosounder data) will enable exploration of potential signatures of gradients in trawling disturbance on benthic habitats and communities.

Objective 2: Benthic habitats and communities of the central crest region of Chatham Rise.

Work on this objective was completed in 2014, with detailed results and conclusions presented in NIWA Client Reports WLG2012-25 (Rowden et al. 2013) and WLG2014-9 (Rowden et al. 2014), and summarised in two progress reports to MPI.

Data on benthic epifaunal communities and physical substratum type extracted from the analysis of 937 seabed photographs from voyage TAN1306 were combined with comparable data from 3908 photographs collected by CRP Ltd during a commercial survey of the area in 2012 (RV *Dorado Discovery*) and subsequently analysed by NIWA, to generate a comprehensive dataset for the central Chatham Rise. Distribution maps of observed occurrences were generated for individual benthic taxa and for eight main benthic community types identified using multivariate statistical techniques. In addition to maps of observed point-occurrence, continuous coverage maps for individual taxa and communities were generated using the predictive species-environment modelling technique Boosted Regression Trees (BRT), which uses the relationships between observed point occurrences and gradients in physical environmental variables to predict probabilities of suitable habitat existing in unsampled areas.

Point-occurrence data showed that substrata across most of the crest of Chatham Rise, including the BPA, were primarily muddy soft sediments with sparse epifauna. The exception to this was in the central western part of the BPA, between approximately 179° 00’ E and 179° 40’ W, where areas of exposed hard substrata in the form of gravel-to-boulder-sized phosphorite rock were widespread. These hard substrata supported patches of diverse epifaunal communities, often characterised by presence of the scleractinian stony coral *Goniocorella dumosa*. All such *G. dumosa* dominated habitats recorded were within the BPA and, with one exception, all were within the CRP Ltd, mining license area (license #50270). Predictive models, however, suggested that suitable environmental conditions for *G. dumosa* might also exist in an area to the north-west of the BPA that has not, to date, been surveyed. An important caveat for these predictions, however, is that the models do not include a detailed substratum-type layer. Because recruitment of *G. dumosa* and other sessile suspension-feeding taxa is dependent on the availability of exposed hard substrata at the seabed (i.e., rock), suitable oceanographic conditions are not sufficient in themselves to predict the occurrence of these taxa with any confidence.

ZBD2014-10 BPA Biodiversity

This project started in July 2015. It has four main objectives that will be addressed sequentially through to 2017:

1) To update the inventory of benthic samples and biodiversity data available within Benthic Protection Areas 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.

The project builds on initial work reported by Clark et al. (2014) that summarised NIWA data on benthic invertebrate samples from BPAs and Seamount Closure Areas.

Under Objective 1, a new extract of specimen data from the NIWA Invertebrate Collection has been combined with additional data from Auckland Museum and Te Papa Tongarewa collections. Within BPAs there have been 1467 stations with almost 15 000 faunal records. Seamount Closure Areas have 340 stations with 3300 faunal records. Data reports have been provided to MPI and presented to the Biodiversity Research Advisory Group, which summarise for each BPA or Seamount area the number of tows, depth ranges, number of records by phylum, and their level of identification (to family, genus or species).

Objective 2 has identified samples that have been collected but not examined, and priority areas where maximum benefit can be derived from processing further material. In conjunction with the Department of Conservation, samples are being identified by New Zealand taxonomists for Campbell Heritage and Campbell East BPAs. This will be expanded if possible to other subantarctic areas.

The additional records will then support Objective 4 in 2017 to evaluate the effectiveness of existing BPAs.

ZBD2016-11 Quantifying benthic biodiversity across natural gradients (new project)

Quantitative data on the distribution and abundance of benthic species are sparse in New Zealand waters. This situation has resulted in high levels of uncertainty opportunities both for limited field validation of existing models (e.g., those developed from the Chatham-Challenger OS20/20 surveys), and development of new abundance-based models. These new models will have improved predictive ability that will better inform management options, and the data and models generated by the project will also inform a benthic risk assessment being developed under a separate MPI project (BEN2014-01).

Other research relevant or specifically linked to the projects above, is listed in Table 18.2.

Table 18.2: Other research linked to Objective 1 habitat classification and characterisation.

MPI	HAB2007-01 Biogenic habitats as areas of particular significance for fisheries management (complete) ZBD2006-02 NABIS (ongoing) Useful data related to defining potential VMEs are collected by MPI scientific fisheries observers working on NZ authorised fishing vessels that operate on the high seas in the South Pacific.
CRI Marine platform or MBIE funding	NIWA Coasts & Oceans centre core-funded programmes: Programme 4 - Marine ecosystem structure and function Programme 6 - Marine biosecurity C01X0907 Coastal Conservation Management (fish habitat classification) CO1X0906 Vulnerable deepsea communities (mapping and sampling a range of deepsea habitats (seamounts, slope, canyons, seeps, vents)) (NIWA)
DOC	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. Ongoing project that is developing a biophysical habitat classification system based on the JNCC classification system for the NZ coastal marine environment. DOC – currently developing a biological habitat classification system for the NZ coastal environment, along with methodologies for habitat mapping as part of a monitoring ‘toolbox’.
OTHER	Victoria University of Wellington – ongoing projects involving marine biodiversity identification; marine protected areas.
EMERGING ISSUES	What portion of a given habitat type should remain intact to support sustainable ecosystems? What are the most effective predictive tools for predicting biodiversity in areas as yet unsampled?

18.3.3 PROGRESS ON SCIENCE OBJECTIVE 2. ECOSYSTEM-SCALE RESEARCH

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 to 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 deepwater 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 fishstocks under various harvesting strategies. It is also important that modellers 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 variable), and wasp-waisted systems where there are major effects in both directions.

PROJECTS

ZBD2008-01 Inshore biogenic habitats

Existing knowledge on biogenic habitat-formers in the <5–200 m depth zone of New Zealand's continental shelf, from sources including structured fisher interviews ('Local Ecological Knowledge', LEK), primary and grey literature, and other sources have been integrated to generate maps of key biogenic habitats in New Zealand coastal waters.

Over 600 targets of interest were identified and marked on marine charts, with more than 200 of these targets being biogenic in nature. Fieldwork has been completed to verify and quantify biodiversity in biogenic habitats using Ocean Survey 20/20 vessel days on *Tangaroa* and a new MSI project to extend the survey potential of the project. New biogenic habitats have been identified, including extensive worm tube 'meadows' off the east coast of the South Island ('the Hay Paddock' and 'Wire-weed'), with associated relatively high epi-faunal invertebrate diversity compared to adjacent bare sediments. Over 60 new species were also collected (dominated by sponges), along with range extensions of many other species. Analyses are underway for key selected areas included in the *Tangaroa* voyages, including offshore North Taranaki Bight, Ranfurly Bank, the polychaete meadows mentioned above, and the Otago Peninsula bryozoan fields.

ZBD2014-07 Southern Coralline algae habitat

Coralline algae are a structurally important component of coastal habitats, and play an important role in ecosystem processes. This project has the following objectives:

1. Document critical baseline information on the diversity of coralline algae in southern New Zealand using morphological and molecular identification.
2. Develop coralline reference collection from habitats and regions predicted to experience stress from ocean acidification.
3. Prepare identification guide for coralline algae of southern New Zealand.

ZBD2014-06 Macroalgae mapping and utility as national indicators

Kelp and fucoid algae form large underwater forests, providing three dimensional structures for fish and invertebrate species and creating canopy for other algae.

Because of their central role in a range of ecological processes on temperate reefs and adjacent habitats, loss of canopy-forming algae is likely to be associated with significant loss of associated species and ecological function. Loss of subtidal macroalgal forests is an increasingly common problem in temperate marine ecosystems, particularly on urbanised coasts. This project has the following objectives:

1. Summarise:
    a) the national and international literature on the use of macroalgae in monitoring programmes,
    b) data available for laminarians and fucoid algae in New Zealand with respect to use in monitoring programmes, distributional data and response to environmental change.
2. Establish the susceptibility of selected New Zealand laminarians and fucoids to selected environmental stressors.
3. Establish the utility of selected macroalgae as monitoring tools for the assessment of ecosystem health and test methods for mapping distribution for baseline monitoring.

DOC-funded research

Ecological Integrity: This programme, undertaken through a partnership with Air New Zealand, has an objective to better understand the concept of ecological integrity in the New Zealand marine environment, and develop a suite of effective and comprehensive methods and tools for monitoring and reporting on species and ecosystems, processes, functioning and health in the marine environment. A key area of research and development work has been the identification and testing of indicators of ecological integrity for New Zealand's marine protected areas, but the application of these indicators may extend beyond these conservation areas. The indicators are part of a broader Biodiversity Assessment Framework being developed by DOC. An example of one of the indicators being trialed is the use of functional traits (de Juan et al. 2015).

Southeast Marine Protection Forum: DOC is supporting a stakeholder-led forum that is developing MPA proposals for the Otago coast, including provision of technical and scientific support (such as provision of SeaSketch as a conservation planning tool).

Review of citizen science programmes for marine monitoring: DOC is exploring mechanisms for better involving communities in marine monitoring and in turn increasing the capacity to undertake monitoring in the marine environment.

Estuaries: DOC is developing an internet-based estuaries resource hub. The first component connects current estuarine restoration and monitoring work around the country and provides restoration and citizen science resources (some are specifically for iwi, hapū and whānau). The second part will be a GIS portal with management relevant information layers designed for a wide range of users.

Marine ecosystem services: A review of ecosystem services in the marine environment has been completed. [57] Additional work is ongoing on developing an ecosystem services matrix. The objective of that work is to produce a methodology for characterising the benefits provided by different ecosystem components within the coastal marine environment.

Cultural indicators programme: DOC is supporting the development of Marine Cultural Health Indicators with Te Rūnanga o Toa Rangatira (TRoTR) and Greater Wellington Regional Council (GWRC). This work will deliver a recommended monitoring programme that can be utilised by DOC, TRoTR and GWRC to monitor the marine environment from a Ngāti Toa perspective; a way in which TRoTR can build capability of staff and iwi members; a pathway to re-connecting the iwi back to the moana; and a holistic approach to monitoring.

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[57] van den Belt, M; Cole, A (2014) Ecosystem goods and services in marine protected areas (MPAs). *Science for Conservation 326*. DOC. Retrieved from http://www.doc.govt.nz/upload/documents/science-and-technical/sfc326entire.pdf.

Ecologically Significant Marine Areas: The objective of this research programme is to identify ecologically significant areas (ESMAs) within New Zealand's marine environment (including the Territorial Sea and EEZ). Work to date includes a 'think piece' report including a review of methodological approach for modelling ESMAs.

Near-shore 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. MPI BioInfo research seeks to help disentangle this complexity, recognising that there will be contributions to this from both biodiversity research and Fisheries Services research.

Regional Councils and universities support some research projects and survey programmes in coastal and estuarine waters by investigating the effects of sedimentation, pollution, ocean outfalls, sand dredge spoils, sand mining and nutrient enrichment on the marine ecosystem.[58] Although this workstream applies to offshore areas as well as near-shore, research to date has focused on the near-shore.

Other research relevant or specifically linked to the projects above, is listed in Table 18.3.

Table 18.3: Other research linked to ecosystem-scale understanding of biodiversity in the marine environment. [Continued on next page]

MPI	ENV2006-04 Ecosystem indicators for New Zealand fisheries ENV2007-04 Climate and oceanographic trends relevant to New Zealand fisheries ENV2007-06 Trophic relationships of commercial middle depth species on the Chatham Rise ANT2015-01, ANT2016-01 – many objectives concerning ecosystem effects of fishing in the Ross Sea region, including research on spatial modelling of populations, multispecies minimum realistic modelling and biology/ecology of bycatch species DEE2010-05 Environmental Indicators for Deepwater Fisheries (Tuck et al. 2014) ZBD2005-05 Taking stock project ZBD2012-02 Tier 1 Statistic (Oceans) ZBD2008-15 Continuous Plankton Recorder – See Robinson et al. (2014). [and subsequent CPR project]

[58] See MFish Biodiversity Research Programme 2010: Part 4. Reference Materials and Other Research.

Table 18.3 [Continued]:

OTHER	AUT: Deepsea and subtidal food web dynamics; offshore & coastal biodiversity postgraduate studies. MBIE contestable: C01X1001 'Protecting Ross Sea Ecosystems' – ecosystem effects of fishing in the Ross Sea region. Completed refer to MPI contract. MBIE contestable: C01X1226 'Ross Sea Climate and Ecosystems' – effects of climate variability/change on the ecosystems of the Ross Sea region. Ongoing. NZARI: 'Top predators of the Ross Sea' – Whales, seals and penguins in the Ross Sea region to understand fishery-predator interactions (http://nzari.aq/nzari-funded-projects#p2013-06). Extended and ongoing (links to MPI project). MBIE contestable: 'Marine Futures' – ecosystem modelling and management in the Hauraki Gulf (http://www.niwa.co.nz/coasts-and-oceans/research-projects/marine-futures). Complete, mapped into challenge. MBIE National Science Challenge: 'Sustainable Seas' (http://www.mbie.govt.nz/info-services/science-innovation/national-science-challenges/sustainable-seas). Waikato University: Environmental Research Unit, Coastal and Marine Ecosystems (http://www.waikato.ac.nz/eri/research/coastal-and-marine-ecosystems). Otago University: Department of Marine Science (http://www.otago.ac.nz/marinescience). Victoria University of Wellington: Ongoing projects involving marine biodiversity identification; marine protected areas; fisheries, including stock assessment, in an EBM framework; work on global climate change, including OA; biosecurity; rock lobster connectivity. University of Auckland: Ongoing research on novel methods to measure marine biodiversity, coastal genetics studies, and marine mammal research. Centre of Research Excellence: Te Pūnaha Matatini (Centre for Complex Systems and Networks) including a project on modelling biological and economic values of marine food resources (http://www.science.auckland.ac.nz/en/about/our-research/research-in-the-faculty-of-science/te-punaha-matatini.html). SeaChange: Hauraki Gulf Marine Spatial Plan (http://www.seachange.org.nz). DOC research on marine ecological integrity? Ongoing, suite of recommended indicators and accompanying monitoring techniques.

PROJECTS

ZBD2014-03 Sub-lethal effects of environment change on fish populations

This project (co-funded with MBIE) will investigate the potential effects of ocean acidification on New Zealand's fish and fisheries, with a focus on snapper. In 2015–16 the project team hosted a workshop of key collaborators, and this year the team have initiated 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 will be conducted to assess the response (e.g., mortality, morphology, energy utilisation, behaviour) of snapper larvae to different acidification scenarios. Finally, the findings of the review and tank experiments will be combined in a deterministic model to assess the effects of acidification at a broader population level

DOC research on functional trait diversity

(see also DOC contract report, Hewitt et al. 2014)

As part of DOC's Ecological Integrity Programme, the use of a traits-based functional approach to the analysis of video imagery was explored. Functional traits that could be determined from video were derived from international literature and tested using video data collected by DOC. Six broad functional categories were used (living position, growth form, body flexibility, mobility, feeding mode, and size; these represent traits that are important for vulnerability, resilience, recovery as well as aspects of ecosystem functioning). A Biological Traits Analysis (BTA) supplemented by estimates of spatial heterogeneity (habitat transitions) and vertical habitat complexity was used to determine functional integrity. BTA fulfil most of the requirements of a good bio-monitoring tool, being well rooted in ecological theory, demonstrated to show responses to changes in environmental conditions and human disturbances and stability across regional species pools and time, and are directly and indirectly related to ecological functions and ecosystem goods and services. This project demonstrated the first step to an index of ecological integrity by successfully converting video data to functional traits data, in a way expected to be habitat independent.

Other research relevant or specifically linked to the projects above, are listed in Table 18.4.

Table 18.4: Other research linked to investigation of the role of biodiversity in the functional ecology of near-shore and offshore marine communities.

MPI	BEN2007-01 Assessing the effects of fishing on soft sediment habitat, fauna, and processes ongoing. HAB2007-01 Biogenic habitats as areas of particular significance for fisheries management complete. ZBD201202 Tier 1 Statistic Oceans. ZBD201302 VME – Genetic Connectivity. ZBD201303 Continuous Plankton Recorder-2.
CRI Marine Platform	C01X1005 Management of Cumulative Effects of Stressors on Aquatic Ecosystems. Ongoing CO1X0907 Coastal Conservation Management, Freshwater and Estuaries and Coasts and Oceans
DOC	EI functional trait project
MPI Biosecurity	Biosecurity surveys: Every 6 months marine high-risk site surveillance.
OTHER	Universities; National Science Challenge
EMERGING ISSUES	Cumulative footprint of human activities; understanding cumulative impacts and risks; marine spatial planning. Land-based effects on marine biodiversity and inshore/offshore habitats. Ecosystem-based management and integrative governance. Science challenge (http://www.sustainableseaschallenge.co.nz). Defining marine ecosystem services, linking them to ecosystem function and societal values.

18.3.4 PROGRESS ON SCIENCE OBJECTIVE 4. MARINE GENETIC BIODIVERSITY

Genetic biodiversity can be measured directly at the scale of genes and chromosomes or indirectly by measuring physical features at the organism scale (assuming that 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 so on.

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

ZBD2009-10 Multi-species analysis of coastal marine connectivity

Following the completion of an extensive literature review of published and unpublished information and the identification of gaps in knowledge about taxa, habitats and spatial coverage of sampling, research focused on the development of a standardised collecting protocol and the development and application of microsatellite markers to quantify the population genetic structure and the coastal connectivity of these taxa (Gardner et al. 2010). Open sandy shores and estuarine environments were highlighted as needing attention. For this, two PhD students carried out field work, genetic analyses, and interpretation of patterns of genetic population structure and the identification of barriers to gene flow in two species of shellfish (tuatua and pipi) and two species of flatfish (yellow-bellied flounder and sand flounder). Both PhD theses are now complete (examined, revised and submitted to the VUW library). Work is currently underway, in conjunction with the scallop work described below, to complete the writing of an Aquatic Environment and Biodiversity Report, and to compile an integrated library of references for submission to MPI. Further publications from the two PhD theses are also in preparation.

The coastal connectivity project has been extended to incorporate a new component of coastal connectivity, with work on the New Zealand scallop, *Pecten novaezelandiae*. This work focuses on population genetic structure and genetic connectivity at two different spatial scales and uses microsatellite markers (consistent with the study already concluded for the four species of the original).

First, the extension work focuses on scallops across New Zealand (the full range of this species' distribution). Samples have been sourced from several regions including the fiords, the far north, and central New Zealand. Genetic analyses and writing up (in the form of a PhD thesis chapter) have been done and this study is now complete. Second, the extension work focuses on scallops in the Hauraki Gulf and Coromandel Peninsula region where an important fishery exists. Scallops have been collected from several populations in this region and genetic connectivity is being assessed to determine linkages among populations at small spatial and temporal scales. This information will be of particular relevance to support management of the Coromandel scallop fishery.

ZBD2013-02 Vulnerable marine ecosystems (VME) genetic connectivity

VMEs are ecosystems comprising species, communities and/or habitats that are highly vulnerable to disturbance, yet little is known about the distribution of biodiversity or genetic relationships within and between VMEs in the deepseas surrounding New Zealand. This project addresses the critical lack of data concerning deepsea 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.

One postdoctoral research fellow and one research assistant joined the project (April 2014). Following systematic examination of specimens preserved in the

NIWA Invertebrate Collection (NIC), five species in two VME orders were identified as having sufficient representation over a wide geographic range within and beyond New Zealand's EEZ. These species are *Enallopsammia rostrata*, *Desmophyllum dianthus* (Order: Scleractinia, stony corals), *Leipoathes secunda*, *Bathypathes patula* and *Stichopathes variabilis* (Order: Antipatharia, black corals). Scleractinian specimens are by far the most numerous in the NIC, and the RV *Tangaroa* TAN1402 cruise to the Louisville Ridge (Jan–Mar 2014) yielded prolific collections of *D. dianthus* in particular. The scleractinian species choice complements ongoing VME genetic connectivity work at Victoria University of Wellington, and data from both projects will provide the most comprehensive dataset on deepsea coral connectivity in New Zealand. Black corals are globally abundant yet poorly understood deepsea invertebrates, and are all protected in New Zealand. Several specimens of *Bathypathes patula* were collected in Antarctica, therefore connectivity between New Zealand and this area may also be inferred. Furthermore, *Enallopsammia rostrata*, *D. dianthus* and *B. patula* are cosmopolitan species and data from this research will be of interest to the wider deepsea community.

Genetic markers are currently being optimised for each species and include a combination of DNA sequences from various genes, microsatellites and single nucleotide polymorphisms (SNPs). Preliminary genetic analyses suggest a lack of concordant connectivity patterns between species, indicate potential genetic isolation in Louisville, and have identified potential cryptic speciation within *L. secunda*. Genetic analyses will continue into 2015, and data will be incorporated into a hydrodynamic modelling framework during the next stage of the South Pacific VME project. Octocorals are also omnipresent in deepsea communities and are well represented in the NIC, albeit with poor species-level identifications. In 2015, genetic connectivity analyses on octocorals will commence, providing much needed data for another abundant, poorly understood and highly vulnerable deepsea taxon.

Other research relevant or specifically linked to the projects above, are listed in Table 18.5.

Table 18.5: Other research linked to marine genetic biodiversity.

MPI	ENH2007-01 Stock enhancement of blackfoot paua. GEN2007-01 Genetic population profile of blackfoot paua. ENH2007-02 Outbreeding depression in invertebrate populations. IPY2007-01 Objective 11. Barcode of life.
NIWA core funding	*Marine Biosecurity.* NIWA Coasts & Oceans core funded Programme 6: Identifying and evaluating biosecurity threats to marine ecosystems from non-indigenous species and developing tools and approaches to prevent entry, reduce establishment and mitigate impacts. Programme includes Cawthron development of molecular tools for identification of non-indigenous species.
OTHER	Universities
EMERGING ISSUES	Can genetics combined with hydrographic models usefully contribute to the identification of biodiversity hot-spots and/or to source-sink relationships within ecosystems?

18.3.5 PROGRESS ON SCIENCE OBJECTIVE 5. EFFECTS OF CLIMATE CHANGE AND VARIABILITY ON MARINE BIODIVERSITY

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, which resulted in a warmer ocean around New Zealand from 1996–2002, can have flow-on effects on ecosystem functioning.

A new report was launched in 2010 by the United Nations on ocean acidification.[59] Among other findings, the study shows that increasing ocean acidification will mean that by 2100 some 70% of cold water corals, (a key refuge and feeding ground for some commercial fish species), will be exposed to corrosive waters (see also Tracey et al. 2011b). 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.

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.

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[60] and has been identified as a gap in biodiversity research in New Zealand.[61] A refresh of the New Zealand Biodiversity Strategy is underway by DOC and will include a chapter on climate change.

PROJECTS

Completed

ZBD2008-11 Predicting plankton biodiversity & productivity with ocean acidification.

This five-year project has provided new information on the potential impacts of ocean acidification and warming on future plankton abundance, distribution, and biodiversity. The study focused primarily on two phytoplankton groups, the coccolithophores, which are present throughout New Zealand waters and form large blooms in productive frontal regions, and the nitrogen-fixing diazotrophs, which are important in nutrient-depleted (oligotrophic) subtropical waters, north of New Zealand.

The project also considered two other potentially susceptible, carbonate-forming zooplankton groups, the Foraminifera and Pteropoda. The project has used observational data from research voyages, including time-series and remote-sensing data, to establish the species community composition and abundance for each group, and generate a present-day baseline against which future change can be assessed. In addition, comparison with environmental variables has identified the key drivers of coccolithophore and diazotroph distribution and abundance. Manipulation experiments conducted in the laboratory and at sea to assess the sensitivity of extant plankton species to future changes, have established the susceptibility of diazotrophs and coccolithophores to future reductions in pH, both alone and in combination with projected increases in temperature. Together the results provide novel insight into the current and potential future status of important planktonic groups in New Zealand waters.

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[59] UN News Centre. Growing ocean acidification threatens marine life, says UN-backed report. 2 December 2010. Retrieved from http://www.un.org/apps/news/story.asp?NewsID=36941&Cr=emissions&Cr1#.WjMlhUqWaUk.

[60] IISD. UNGA's Second Committee Considers Biodiversity and Sustainable Development. Retrieved from http://biodiversity-l.iisd.org/news/ungas-second-committee-considers-biodiversity-and-sustainable-development.

[61] Green, W; Clarkson, B (2006) Review of the New Zealand Biodiversity Strategy Themes. Retrieved from http://www.doc.govt.nz/Documents/conservation/nz-biodiversity-strategy-themes.pdf.

A regional survey provided a baseline dataset of coccolithophore diversity, abundance and biogeography for New Zealand surface waters, with 46 species identified, of which 31 were new or recently identified species, and one a previously undescribed species of *Syracosphaera*. The highest coccolithophore abundance occurred in the region of the Subtropical Front on the Chatham Rise, with coccolithophores accounting for 30% of carbonate and 26% of algal particulate organic carbon in surface waters during summer. Coccolithophore species diversity was greatest in the frontal regions on the Chatham Rise and in the southern Tasman Sea, but with significant variation in the dominant species observed over relatively small spatial scales. The globally ubiquitous coccolithophore, *Emiliania huxleyi*, was the dominant species in both bloom and non-bloom conditions, often exceeding cell abundances of 1 million cells per litre, 1–2 orders of magnitude greater than any other coccolithophore species. The observed restriction of elevated cell abundances of *E. huxleyi* to surface waters of low silicate concentration indicated that this species may be outcompeted by diatoms at higher silicate concentrations, and hence the growth and distribution of diatoms may restrict *E. huxleyi* blooms. *E. huxleyi* and the second-most dominant coccolithophore species, *Reticulofenestra parvula*, showed contrasting relationships with temperature and macronutrients, with *E. huxleyi* associated with warmer water of low silicate concentration and low N:P. Comparison with temperatures and nutrients projected for the Chatham Rise region by 2100 suggest that *E. huxleyi* may be the more successful of these two species in the future in this region. The inferred future increase in *E. huxleyi* blooms would influence both productivity and CO_2 uptake on the Chatham Rise, and so it is recommended that the role and contribution of coccolithophores in both the pelagic food web and vertical carbon export be established in this region.

The unique light-scattering properties of the coccoliths (carbonate plates) of *E. huxleyi* provides potential for quantifying this carbonate and the spatial distribution of this species in surface waters by satellite remote-sensing. An optical algorithm developed for other ocean regions was successfully tested and validated for New Zealand waters, by comparison of satellite-derived carbonate with *in situ* carbonate concentrations. The algorithm was then applied to ocean colour images, from the SeaWifs and MODIS satellites, for the period 1997–2014 to produce a Coccolithophore Atlas of seasonal climatologies and inter-annual trends in surface water carbonate associated with *E. huxleyi*. Blooms of *E. huxleyi* were identified around much of New Zealand, with the satellite data confirming shipboard observations that the Chatham Rise supported the highest carbonate concentrations and presence of *E. huxleyi*. The Atlas showed a southerly seasonal progression in bloom location, with a surface carbonate maximum in spring in subtropical waters, in mid-summer in the Subtropical Front, and in late summer in subantarctic waters. The time-series record also showed a low, but statistically significant, increase in carbonate between 42 and 47^{o}S, suggestive of a minor increase in *E. huxleyi* abundance over the 15-year period in the Subtropical Front and northern subantarctic waters. A second remote-sensing approach was developed for assessing *E. huxleyi* bloom frequency that also indicated an increase in the Chatham Rise region, but also identified an increase in bloom frequency in northern subtropical waters, which was not apparent from the satellite carbonate algorithm. The absence of a temporal trend in satellite-derived carbonate in surface subantarctic waters in the region of the Munida time-series off the Otago coast, where pH has been decreasing since 2000, suggests that ocean acidification is not currently a primary determinant of *E. huxleyi* abundance in these waters. Conversely, comparison of sea surface temperature and carbonate anomalies derived from satellite data indicated a positive relationship in subantarctic waters and a negative relationship in subtropical waters, suggesting that warming may be a more important determinant of *E. huxleyi* abundance. These opposing relationships with temperature may be attributable to the differing impacts of increased stratification and associated reductions in nutrient supply, and suggests contrasting future responses in coccolithophore abundance in the two main water masses around New Zealand. This study validated the application of remote-sensing for determining *E. huxleyi* distribution, and provided a baseline against which future changes in *E. huxleyi* distribution and biomass in New Zealand surface waters can be assessed. Application of this approach to monitor changes in *E. huxleyi* distribution and abundance would benefit from further surface sampling of carbonate and *E. huxleyi,* and validation, in different oceanic regions around New Zealand.

Manipulation experiments were carried out on unispecific cultures of *E. huxleyi* in the laboratory, and also on mixed natural populations containing coccolithophores at sea, to determine the individual and combined impact of temperature and ocean acidification on coccolithophore abundance, morphology and carbon content. Unispecific cultures of *E. huxleyi* showed no sensitivity across a range of CO_2 concentrations in terms of growth rate and particulate organic carbon production over 16 days, although a significant decrease in calcification rate was observed. Comparison of these results with reported responses in different strains of *E. huxleyi*, including from New Zealand waters, showed considerable variability in the response of growth and particulate organic carbon to elevated CO_2, whereas a decline in the carbonate:particulate organic carbon ratio was a consistent response under high CO_2 conditions. The implications of this are currently unclear, but this may influence the survival of *E. huxleyi* in the future ocean, and also the transfer of organic carbon and carbonate into the deep ocean since carbonate acts as 'ballast' and enhances particle export from the surface ocean. In the manipulation experiments at sea using natural mixed plankton communities, *E. huxleyi* abundance was not significantly altered when exposed to elevated CO_2 (750 ppmv) alone, and in combination with elevated temperature (+3–4°C). Overall, the laboratory and shipboard results suggest that *E. huxleyi* abundance may not change significantly in response to changes in CO_2 projected for 2100, despite an observed decline in particulate inorganic to organic carbon ratios under elevated CO_2 conditions. From a methodological viewpoint, the present study highlights the importance of considering the effects of interactions of climate-related stressors, and also in using complementary unispecific and mixed community experiments. Further natural community experiments are required to determine the net effects of future changes in temperature and CO_2 on the food quality of coccolithophores and other phytoplankton groups to zooplankton, to establish the potential impacts of climate change on food web dynamics and productivity. In particular, it would be valuable to determine how potential changes in the carbonate to particulate organic carbon ratio of *E. huxleyi* may influence grazing and the supply of carbon to the deep ocean.

A semi-continuous, 11-year record of the abundance of two carbonate-forming zooplankton groups, the foraminifera and pteropods, was obtained in the two major water masses to establish a biodiversity baseline in New Zealand waters, and determine whether significant changes in abundance occurred during the period 2000–12. The samples were obtained from sediment traps deployed at 1500 m depth on two time-series moorings, one in subtropical water and one in subantarctic water. In general, the foraminiferal fluxes showed a close association with calcium carbonate fluxes, with total fluxes in subantarctic water exceeding those in subtropical water, whereas the reverse was observed for organic carbon fluxes. Highest foraminiferal fluxes occurred in spring and autumn in subantarctic waters, and in summer (or occasionally winter) in subtropical water. Significant inter-annual variability was observed, and there was no significant inter-annual trend in foraminiferal abundance over the 11-year time-series in either water mass. However, there was evidence of a reduction in total foraminifera flux from 2008–12, and an increase in more tropical species towards the end of the time-series. Pteropod flux also showed large inter-annual variability, with considerably higher abundances in subtropical waters where abundance was dominated by a single species (*Limacina inflata*). Experiments on pteropods tests showed no evidence of carbonate dissolution at the under-saturated carbonate levels at 1500 m water depth. In contrast to trends reported in a subantarctic water time-series south of Tasmania, there was no evidence of a decline in pteropod abundance in New Zealand subantarctic waters. This suggests that there is currently no effect of ocean acidification on pteropods; however, projected future decreases in the saturation state of aragonite, the form of carbonate used by pteropods to form their shells, may result in a decline in pteropod abundance in these waters. Comparison of pteropod abundance with climatologies of environmental variables suggest that other factors, such as food supply, may be important determinants of pteropod abundance. Establishing the spatial distribution and the ecological role of pteropods and foraminifera in New Zealand waters in new surveys using the Continuous Plankton Recorder would be valuable, as would experimental determination of the sensitivity of the potentially more susceptible juvenile pteropod stages to ocean acidification.

The distribution of nitrogen fixation rate and the nitrogen-fixing diazotrophs were established in subtropical waters north of New Zealand, by determining the presence of the *nifh* gene that encodes for the enzyme used in nitrogen fixation. Four different diazotroph phylotypes were identified in samples from three research voyages in the Tasman Sea, with the highest diversity and abundance coincident with highest nitrogen fixation rates both within, and north of, the Tasman Front. There was a corresponding decline in diazotroph abundance and nitrogen fixation in the southern Tasman Sea, coincident with decreasing surface seawater temperature. The different phylotypes were generally closely associated, with higher abundances, particularly for the large colonial *Trichodesmium* sp., north of the Tasman Front. The smaller Unicellular Cyanobacteria A (UCYN A) phylotype was the dominant diazotroph in the Tasman Sea, and nitrogen fixation rate correlated with both total *nifH* abundance, and abundances of all the diazotroph phylotypes, except *Trichodesmium*. North of the Tasman Front, nitrogen fixation contributes less than 15% of total new nitrogen supply, although both the rate, and its proportional contribution to new nitrogen supply, may vary, as was observed during a tropical cyclone. Nitrogen fixation rates during winter in the ultra-oligotrophic surface waters of the South Pacific Gyre, north-east of New Zealand, were near analytical detection limits, and two orders of magnitude below that of the Tasman Sea. These low rates were accompanied by a corresponding absence of the main diazotroph phylotypes, with only gamma(γ)-proteobacteria present. Temperature and phosphate availability (N:P) were the primary environmental variables determining nitrogen fixation rate and diazotroph abundance in the Tasman Sea. Nitrogen fixation was detectable down to 14°C, with most of the diazotroph phylotypes recorded at lower temperatures than reported for the Atlantic; however nitrogen fixation rates were only significant (>1 nmol/l/d) at temperatures exceeding 19.5°C. As the Tasman Sea region is experiencing rates of warming that exceed the global average, this suggests that diazotroph abundance and nitrogen fixation may increase in the future. With projected further warming of surface waters and associated increases in stratification, nitrogen fixation may become a more significant source of nutrients for primary production in New Zealand waters, and so it is important to establish how nitrogen fixed by diazotrophs is transferred into the marine food web.

A number of studies have shown that the diazotroph species *Trichodesmium* may benefit from ocean acidification, as it increases carbon and nitrogen fixation under elevated CO_2. Manipulation experiments were carried out on natural mixed plankton communities containing nitrogen fixers from subtropical waters north-east and north-west of New Zealand, under elevated CO_2 alone (750 ppmv), and in combination with elevated temperature (+3°C). Contrary to observations in other regions, nitrogen fixation was not stimulated by either elevated CO_2, or in combination with elevated temperature, at any of the four sampling sites. This most likely reflects that the diazotroph community was dominated by UCYN A and gamma-proteobacteria, and not the larger filamentous *Trichodesmium* species. As these smaller diazotrophs are heterotrophs they may gain little benefit from an increase in dissolved CO_2; however, the observed response may reflect other factors, such as the larger cell surface area:volume of small diazotrophs, the potential for iron limitation of the response to elevated CO_2, or that another component of the microbial community outcompeted the diazotrophs. Regardless, the results highlight the importance of studying endemic species and groups, and using natural mixed communities, to determine how climate change and ocean acidification will influence plankton biodiversity and productivity in New Zealand waters.

ZBD2014-01 Age and growth study of deepsea coral in aquaria

Research funding has been provided to improve our understanding of the impacts of ocean acidification on deepsea coral growth. An initial study was carried out to evaluate the feasibility of successfully collecting live specimens at sea and maintaining deepsea corals in the laboratory. Under Ministry for Primary Industries (MPI) Project SEA2013-01, a healthy colony of the deepsea stony branching coral *Solenosmilia variabilis* was successfully maintained under laboratory conditions at NIWA for 14 months, demonstrating the possibility of carrying out future experimental work with this species. NIWA subsequently entered into an agreement with MPI to extend work on deepsea corals (Project ZBD201401). New coral colonies, comprising several live polyps per colony, were collected in March 2014 during the High Seas *Tangaroa* voyage to the Louisville Ridge (for the purpose of carrying out multiple stressor experiments in aquaria). With the limited funding available, the experiment is focused on ocean acidification only. The project includes a collaboration between NIWA and Victoria University School of Biological Sciences postgraduate programme with involvement of a Master's student in the in-aquaria experiments.

Ocean acidification manipulation is now underway to look at the physiological responses (e.g., growth) to future predicted environmental conditions. The experiment was initiated on 29 July 2014, when coral colonies were divided up and assigned to replicate control (pH 7.88) and treatment (pH 7.65) tanks. The corals are being held in the NIWA OA facility where small pieces from several coral colonies have been held in chambers in the dark, and in optimal temperature 3.5 degrees flow rate based on current velocity data for the region, (flow rate of 120 mL/min and 240 mL/min) turning over the volume of the chambers every 15 minutes. The corals are fed twice a week with KorallFluid and 16 months on the colonies are alive with tentacles extending from many of the individual calyces.

To date radial and linear extension and buoyant weight have been regularly measured to observe changes in morphology and measure growth. Some species of deepsea coral up-regulate their intracellular pH when exposed to acidified conditions. This serves as an adaptive response by increasing the internal carbonate saturation state, alleviating the affect that pH reduction has on the availability of carbonate and ease of calcification. Intracellular pH measurements on *S. variabilis* live polyps were made in January and September 2015 to determine whether this species of coral up-regulate intracellular pH when exposed to acidified conditions. Respiration rates have also been taken examined at six monthly intervals to determine what energetic costs may be associated with up-regulation and calcification under acidified conditions. The experiment is planned to run until 30 June 2016.

The colonies have survived well in their laboratory environment. Tentacles continue to extend from the polyp mouth in several of the calyces. We note, however, that some small colonies appear stressed, there is a loss of external tissue material on some branch regions and some branches are very bleached. Nevertheless, we have now completed six intermediate sampling points and two major sampling points. As expected, to date we have not observed any linear growth of these very slow growing organisms.

ZBD2014-09 Climate change risks and opportunities in the marine environment, New Zealand.

The overall aim of this project is 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 four-year project has just completed its first year, which involved the initial synthesis of available information on CO_2- and climate-induced changes that affect these regions and fisheries. A workshop was convened to share and discuss results with stakeholder, end-user and research communities. Subsequent years will see incorporation of additional information, in consultation with fishers and managers, and development of useful tools with which to disseminate and use this information.

ZBD2013-06 Impacts of environmental change on shell generation and maintenance of important aquaculture species

Ocean acidification is a real and imminent threat to calcifying organisms, including shellfish. A previous study of potential effects of near future conditions on New Zealand paua and flat oysters revealed effects on survival and condition, and suggested that effects on shell generation and/or integrity may have been a contributing factor. In this additional project (ZBD2013-06), detailed analyses of shells of individuals of each species were conducted to determine how the decreased pH/increased temperature modified their shell (i) thickness, (ii) mineralogy and (iii) construction. The analyses showed more significant effects on paua than on flat oysters, likely due to differences in carbonate composition. For paua, we noted a consistent, strong effect of temperature, and disruption of normal growth and shell characteristics at low pH, which was also stronger at lower temperature. The study also showed the importance of even a small (2°C) difference in temperature on growth and shell characteristics, and on influencing the effects of low pH.

ZBD2016-07 Multiple stressors on coastal ecosystems in situ

The structure and function of marine ecosystems is altering in response to Global Environmental Change (climate change effects and ocean acidification), yet we are currently unable to predict what this will mean for the productivity, biodiversity and ecosystem services in New Zealand coastal regions.

Increasing acidity and water temperatures are two major stressors that will influence the future structure and function of coastal ecosystems. To date research has primarily focused on the response of different faunal groups to acidified conditions in isolation; to advance understanding and 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 will be altered, will be carried out as part of the CARIM Project (Coastal Acidification: Rate, Impacts & Management) currently funded by MBIE. This approach is novel (for New Zealand) in that it integrates all responses in the biogeochemistry and the entire plankton community so that the net ecosystem effects can be determined in the wild as opposed to an aquarium or laboratory situation. The mesocosm experiments will determine the impact of projected warming and acidification on the biomass, productivity and community composition of coastal plankton, and identify potential future changes in coastal biogeochemistry and biodiversity. One of the broader aims of CARIM is to combine observations of indirect ecosystem changes, for example in trophic interaction and substrate availability, with direct effects on organismal physiology to determine the overall impacts on key species of ecologically and socio-economic importance. The mesocosm experiments will provide the platform for examining the indirect effects of change in food availability and quality in the lower food web on key species such as paua and green lipped mussel. To maximise the value of these experiments, additional measurements of change in the nutritional value of the plankton, and its transfer through the food web, are being conducted, so that this can be factored into models of future population success. CARIM is already underway, and the experimental phase has begun.

Other research relevant or specifically linked to the projects above, are listed in Table 18.6.

Table 18.6: Other research linked to effects of climate change and variability on marine biodiversity.

MPI	SAM2005-02 Effects of climate on commercial fish abundance. ENV2007-04 Climate and oceanographic trends relevant to New Zealand fisheries.
CRI Marine Platform, MBIE	C01X502 Coasts & Oceans Centre, CARIM Project.
DOC	Baseline surveys; protected deepsea corals (Tracey et al. 2011b, Baird et al. 2013); development of climate change indicators as a component of protected area monitoring.
OTHER	University of Otago and NIWA: Shelf carbonate geochemistry and bryozoans. Geomarine Services: Foraminiferal record of human impact. Regional Council monitoring programmes. NIWA Coast and Ocean core programme. US-NZ Joint Commission Meeting for scientific and technological exchange. Ongoing ocean acidification work and deepsea coral identification
EMERGING ISSUES (this objective)	How does climate change influence marine microbial diversity, species mix and biogeochemical roles? How will harmful toxic algal blooms be affected by warming seas? (e.g., Chang & Mullan 2003, Chang et al. 2003) How will climate change affect primary industries in the sea, and ecosystem services on which industry depends?

18.3.6 PROGRESS ON SCIENCE OBJECTIVE 6. BIODIVERSTIY METRICS AND OTHER INDICATORS FOR MONITORING CHANGE

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 much reduced set of core indicators that relate to the marine environment have been reported on.[62] A new international initiative launched in 2010: 'Biodiversity Indicators Partnership'[63] 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.

[62] MfE. National Environmental Indicators. Retrieved from http://www.mfe.govt.nz/publications/environmental-reporting/environment-new-zealand-2007-chapter-1-environmental-reportin-5.

[63] Biodiversity Indicators Partnership, https://www.bipindicators.net.

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 (see Biodiversity Research Programme 2010: Part 4). 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 series of workshops to determine how best to approach monitoring on a national scale (ZBD2008-14). (One objective of monitoring would be to test the effectiveness of management measures.)

PROJECTS

ZBD2008-15 Continuous Plankton Recorder (CPR) Project: implementation and identification.

Complete.

ZBD2013-03 Continuous Plankton Recorder (CPR)-Phase 2

The overall objective of the Continuous Plankton Recorder (CPR)-Phase 2 project 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.

The current project, ZBD2013-03, continues this annual programme of CPR sampling and is funded for a further five years. 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 trends in the plankton community.

To date, one summer sampling run has been completed (2013–14). Nine CPR runs were carried out between 30 November 2013 and 11 February 2014. The processing of the samples from these runs is well advanced and should be completed before the end of the second year of sampling, which is again due to commence from Nov/Dec 2014. At this stage, the new data is being collated and stored in the Southern Ocean CPR Survey meta-database.

ZBD2014-04 Isoscapes for trophic studies

The main objective of this project is to develop biochemical maps of the South Pacific that then can be used to understand the movements and migrations of marine animals. These maps are based on the spatial patterns of isotope values found at the base of the marine food web in different oceanic regions, driven mainly by the dominant biogeochemical provinces. This approach, based on both measured values from the open ocean and recently developed ocean models predicting these values, is especially powerful in the South Pacific because of the large latitudinal variations. We see higher carbon isotope values in the tropics and consecutively more negative values as you move to higher latitudes, with the most negative ones found in the Southern Ocean. Physical factors (temperature and dissolved CO_2) drive these large-scale patterns, with some localised variations due to increases in primary production. After the baseline maps are established, samples can then be collected from migratory marine animals to determine if their values match those values observed in the region where they were sampled/captured, after correcting for relative trophic position. If the isotope values are different than what you would expect from the region, these individuals would be considered migratory or new immigrants to the region. For example, white sharks biopsied off of Steward Island in March of each year reflect an equatorial or tropical signal, demonstrating not only were the sharks travelling to these regions but they were foraging and fueling their migration back to Stewart Island every year (Graham et al. 2010). Therefore, these chemical tracers can be used as tags to determine the movements of elusive marine animals or those that are difficult to employ other tracking techniques.

In addition to using isoscapes and the isotopes of marine animals to determine the movements of individuals in the open ocean, these techniques can also provide information on their diet and dietary changes in space and time. The information gained on movements and diet are not mutually exclusive. Therefore, when interpreting variations in an animals' isotope values requires careful consideration of both variables. For example, is the marine mammal or seabird's isotope value dissimilar to what we expect from the region of collection/capture or is that individual/group foraging differently than we expected. Incorporating information from other techniques and observations is critical to determine the factors driving the isotope values. In almost all isoscape and chemical tag studies, including those in this project, additional datasets (e.g., genetic, direct observations) and other constraints are required to distinguish the influences of movement and diet on the observed values. We also have another tool, compound-specific isotope analysis of specific amino acids isolated from the tissues of individuals that will indicate if the observed variations are due to diet, movements, or a combination of those two. For example, a recent study examining the diet of humpback whales foraging around the Balleny Islands in the Southern Ocean. Variations were observed in their (bulk) nitrogen isotope values, especially in male humpbacks, which could indicate a difference in diet (more fish) or in migratory pathways all while stopping to 'snack' along the journey. The compound-specific isotope work indicates it is a combination of both – there is trophic position difference in these individuals and there is also an indication they have foraged in areas with different biogeochemical processes (Sarah Bury, NIWA, pers. Comm.). Determining the migratory routes or changes in the diet of the Oceania group of humpbacks in the South Pacific is critical to their conservation, especially as their population growth is not increasing at the same rate as other humpback whale populations in the Pacific (findings from Constantine 2016).

Another excellent example of how isoscapes and the approach of this project can elucidate movements of elusive marine predators in the open ocean is examining the movements of tuna, sharks, and billfish in the Pacific. In the equatorial Pacific, we determined that tuna foraged in a region for a longer period of time than previously understood (traditionally it was assumed all tunas were highly migratory). We could conclude this important finding because the isotope values of the tuna were similar, after correcting for trophic position, to the base of the food web where they were captured by commercial fisheries (Graham et al. 2010; Lorrain et al. 2014). As this project allowed us to develop isoscapes beyond those that we produced for the equatorial Pacific, into the South Pacific, we were able to apply the same approach to examine the movements of tuna, sharks, and billfish around New Zealand. Results strongly indicate that these top predators are more transient (migratory) in New Zealand waters, as very few reflect what we would expect for a New Zealand predator's values. Some species (e.g., mako and blue sharks, southern bluefin tuna), show more migration than other predators and these information is being generated for MPI's highly migratory working group (HMS2010405) and to be incorporated in future stock assessments and management strategies.

Our work continues on using these isoscapes and methods to determine the movements of marine mammals and seabirds in the South Pacific, especially those using the New Zealand EEZ for portions of their lifecycle (e.g., rearing/rookeries, stop overs during migration, and mating). Currently emphasis has been placed on examining the movements of the Oceania population of humpback whales with the considerable effort recently expended on research voyages and biopsing these individuals (e.g., Constantine 2016). It is important to note that the isoscapes and methods employed in this project are of very limited value to coastal ecosystems – the biogeochemistry and terrestrial inputs are too spatially and temporally dynamic to be able to model the coastal isoscapes. Much more detailed, and sampling intensive work is required in coastal systems, and is more likely project-by-project objective driven. However, even with that limitation, the open ocean isoscapes developed and established in this project can be used to examine movements in many marine animals and trophic guilds during current and future studies with reasonable project costs and timelines. This will be especially important to monitor in a changing ocean ecosystem(s).

ZBD214-05 Ocean acidification modelling

The Ocean Acidification Modelling project seeks to determine how much the aragonite and calcite saturation horizons (ASH and CSH) have changed over the industrial era for the South-west Pacific, including New Zealand's EEZ. The project team has collated a wealth of historical observations of temperature, salinity, oxygen and carbonate parameters that have been made between the 1928 and the near present, assessed the data quality, and collated the data. In addition, we gathered output from 10 ocean model simulations that have been optimised against ocean carbon data and used them to construct a time history of anthropogenic carbon storage in the South-west Pacific. The range between models gives an estimate of uncertainty.

In the next phase of the project, the historical data and model simulations will be combined to estimate decadal changes in carbonate parameters, the ASH, and the CSH over the South-west Pacific. This will give us insight into how much adaptation to ocean acidification may have already occurred in the waters around New

Zealand. These data-based estimates of the ASH and CSH will then be compared with simulations from the Coupled Model Intercomparison Project Phase 5 (CMIP5) to explore the implications of these findings for future projections of the ASH and CSH in the South-west Pacific.

Other research relevant or specifically linked to the projects above, are listed in Table 18.7.

Table 18.7: Other research linked to biodiversity metrics and other indicators for monitoring change.

MPI	ENV2006-15 Database and fishing indicator on seamount habitats (Rowden et al. 2008). BEN2009-02 (Tuck et al. 2010). ENV2006-04 Fisheries indicators from trawl surveys (Tuck et al. 2009). DEE2010-04 Development of a methodology for Ecological Risk Assessments for Deepwater Fisheries. DEE2010-05 Development of a suite of ecosystem and environmental indicators for deepwater fisheries. Completed. DEE2010-06 Design a programme to monitor trends in deepwater benthic communities.
CRI Marine Platform, MBIE	Coasts and Oceans Centre
DOC	Development of monitoring and reporting framework for marine protected areas as a component of DOC's Biodiversity Assessment Framework.
OTHER	Regional Councils, Universities Ministry of the Environment Environmental Reporting Act and associated Technical support, Otago University development of the Ocean Acidification Monitoring Network (Kim Currie).
EMERGING ISSUES	Monitoring coastal waters and New Zealand's oceans to report on a national scale remains a major gap that will be addressed in part by the proposed Tier 1 statistic on Oceans and the Environmental Reporting Bill.

18.3.7 SCIENTIFIC OBJECTIVE 7. IDENTIFYING THREATS AND IMPACTS TO BIODIVERSITY AND ECOSYSTEM FUNCTIONING

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. Benthic communities including seamount communities, volcanic vent communities, bryozoans, corals, hydroids and sponges are vulnerable to human disturbance. 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. 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 recent project commissioned by the MPI 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 harbors 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+ assessment completed on Chatham Rise seamounts to estimate the relative risk to seamount benthic habitat from bottom trawling (under ENV2005-16, Clark et al. 2011). An MPI project (DEE2010-04) has resulted in a new ecological risk assessment being developed that is tailored for New Zealand deepwater fisheries.

PROJECTS

ZBD2009-25 Predicting impacts of increasing rates of disturbance on functional diversity in marine benthic ecosystems

This project expanded on a spatially explicit patch dynamic model as a framework to illustrate how increasing rates of disturbance to benthic marine ecosystems influence functional diversity, 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). The aim of the model is to provide a heuristic tool that can be used when considering seafloor disturbance regimes in the context of spatial planning and other ecosystem-based management (Lundquist et al. 2013).

Eight functional groups were defined for the model, representing key aspects of the way organisms in seafloor communities modify their environment and interact with each other. These include: opportunistic early colonists with limited substrate disturbance; opportunistic early colonists with considerable substrate disturbance; substrate stabilisers (e.g., tube mat formers); substrate destabilisers; shell hash-creating species; emergent epifauna; burrowers; and predators and scavengers. While we did not define each functional group as having a sensitivity to disturbance, each functional group is allocated a selection of life history traits based on review of the scientific literature and expert knowledge (i.e., age of maturity, maximum lifespan, seasonality of reproduction, larval dispersal distance).

When disturbance is added, the model predicts changes in the occupancy of functional groups within the model seascape. Response to disturbance and recovery rates differ between the eight functional groups, reflecting the different life history characteristics and dispersal characteristics simulated by the model. Some functional groups respond negatively to disturbance, including those known to be sensitive to, and recover slowly from, disturbance (e.g., emergent epifauna). Other groups (e.g., opportunistic taxa) respond favourably to disturbance in the model, as we would expect.

The model was run to compare with available inshore (Tasman and Golden Bays) and offshore (Chatham Rise and Challenger Plateau) empirical datasets. These datasets included video data with broad coverage of the seafloor, but relatively poor representation of small-bodied and infaunal groups, in combination with benthic sled, grabs or cores that better sampled these groups. We used a fuzzy logic approach based on functional traits (e.g., feeding, motility, position in the sediment, size) to allocate 1056 individual taxonomic units (e.g., species) into one of eight functional groups, and compare relative abundance of functional groups from inshore and offshore surveys to model predictions.

Model predictions were consistent with changes in functional group abundance with increasing rates of disturbance in both the inshore and offshore datasets, with declines in functional group abundance occurring at the approximate disturbance rates predicted by the model. The strong similarity between model and observed community changes with disturbance showcases the value of this heuristic tool, based on fundamental biological parameters, for investigating disturbance and recovery dynamics in seafloor communities. Future research can build on this model framework, varying parameters and assumptions within model scenarios, to inform ecosystem-based management approaches for seafloor communities.

Other research relevant or specifically linked to the projects above, are listed in Table 18.8.

Table 18.8: Other research linked to threats to and impacts on biodiversity. [Continued on next page]

MPI	BEN2007-05 Assessment of anthropogenic threats to New Zealand marine habitats (MacDiarmid et al. 2012). DEE2010-04
MBIE	CO1X0906 Vulnerable deepsea communities (mapping and sampling a range of deepsea habitats (seamounts, slope, canyons, seeps, vents), and determining relative risk to their benthic communities from human activities culminates in risk assessments; megafauna contestable?
MFE	MFE12301 Expert risk assessment of activities in the New Zealand Exclusive Economic Zone and Extended Continental Shelf (MacDiarmid et al. 2011). MFE14301 Environmental risk assessment of discharges of sediment during prospecting and exploration for seabed minerals (MacDiarmid et al. 2014).

Table 18.8 [Continued]:

DOC	Conservation Services Programme, Threat Management Plan, Marine Ecosystems Programme
EMERGING ISSUES	The socio-economic valuation of biodiversity in New Zealand has not been adequately addressed. The cumulative footprint of anthropogenic activities on the New Zealand marine environment has not been assessed. Potential development of seabed mining makes this a priority in deepwater environments as well as coastal.

18.3.8 BIODIVERSITY IN ANTARCTICA: BIOROSS PROJECT SUMMARIES AND PROGRESS

The objectives of BioRoss are to improve understanding of the biodiversity and functional ecology of selected marine communities in the Ross Sea. These objectives are being achieved by commissioning directed research on the diversity and function of selected marine communities in the Ross Sea region. BioRoss is committed to linking with ongoing Ross Sea ecosystems research through the Antarctic Working Group, and supporting climate change related research, especially at high latitudes.

Data acquisition from the Antarctic marine environment is logistically difficult and expensive. Nevertheless, the seven biodiversity Objectives (see Section 18.3.1) also drive BioRoss research projects. The BioRoss survey in 2004 and the Latitudinal Gradient Project ICECUBE have provided significant new information on biodiversity, species abundance and distribution that are now facilitating research into functional ecology and longer-term monitoring programmes. This research has the potential to lead into other research on genetic diversity, climate variability and the development of indicators. The research results are also being used in the MPI Antarctic Research Programme projects on ecosystem modelling of the Ross Sea.

The MPI Antarctic Research and BioRoss Programmes are also directly involved in supporting the development of protection measures around the Balleny Islands. In 2005 MPI scientists and Ministry of Foreign Affairs and Trade (MFAT) personnel prepared a paper for submission to CCAMLR justifying MPA designation around the islands to protect ecosystem processes occurring there that may be important for the stability and function of the wider Ross Sea regional ecosystem.

To collect data in support of the MPA proposal, MPI BioRoss funded a targeted research voyage to the Balleny Islands in February 2006 (ZBD2005-01), and also provided supplementary funding to carry out opportunistic biological sampling at the Balleny Islands on a voyage to the Ross Sea that was primarily funded by LINZ to do bathymetric mapping.

The field sampling of these projects were successful, both providing important data and specimens from the Balleny Islands area and supplementary information for the Antarctic Working Group Research Programme. The results will inform research planning for subsequent projects. Support for Ross Sea region biodiversity will remain a high priority for future research in the BioRoss Programme.

In addition, BioRoss funded a further ICECUBE project to sample the Antarctic coastline during the summer season of 2006–07 (ZBD2006-03). ICECUBE is a key part of the international Latitudinal Gradient Project to explore hypotheses about environmental drivers of structure and function in sub-tidal ecosystems along the western Ross Sea coastline (Cummings et al. 2008). This project acquired funding for three seasons (2007–08, 2008–09, 2009–10) as part of the MBIE IPY contestable round (see also Cummings et al. 2011 and Thrush & Cummings 2011). Published reports and papers from the MPI Ross Sea coastal projects include Cummings et al. 2003, 2006b, 2008, 2010, 2011, De Domenico et al. 2006, Grotti et al. 2008, Guidetti et al. 2006, Norkko et al. 2002, 2004, 2005, 2007, Pinkerton et al. 2006, Schwarz et al. 2003, 2005, Sharp et al. 2010, Sutherland 2008, Thrush et al. 2006, 2010, 2014.

The New Zealand government provided one-off funding for a Census of Antarctic Marine Life (CAML) survey to the Ross Sea from RV *Tangaroa* as part of New Zealand's involvement in the 2007–08 International Polar Year activities. The CAML Voyage was a large cooperative research effort under the banner of Ocean Survey 20/20 with considerable international collaboration, simultaneously utilising a number of different vessels with different strengths and capabilities. Progress on the two projects, IPY2007-01 and IPY2007-02, is detailed below.

PROJECTS

ZBD2013-08 NZ Ross Sea connectivity, humpback whales

The main aims of this research are to determine the breeding ground origins, migratory path, Antarctic feeding grounds and prey of New Zealand's endangered humpback whales. We are using a multidisciplinary approach with satellite telemetry, DNA-based genetic markers, stable isotopes, isoscape modelling and photo-identification all being used to understand individual and population level linkages. The primary objectives are:

i) Deploy satellite tags on humpback whales at Raoul Island to determine the migration pathways and habitat use in Antarctica using spatial analysis tools.

ii) Determine the breeding and feeding ground linkages, site fidelity and levels of mixing through genotype and photo-identification analysis of individual whales.

iii) Determine the isotopic signatures of prey and model the values across the broader Ross Sea region to allow predictive models of humpback stock recovery.

We collected data at Raoul Island from 29 September to 11 October using small research boats deployed from the RV *Braveheart*. During this time we deployed 25 satellite tags, collected 85 tissue samples, and recorded four hours of whale song and photo-identified 129 individuals.

A total of 17 satellite tags have transmitted location information throughout duty cycles of 10 h on and 2 h off. It is not unusual for tags to start transmitting a few weeks after deployment so we may receive signals from other tags over the next few weeks. In general, the whales have migrated in a south-easterly direction with only a few whales coming close to mainland New Zealand. The whales are on course for Antarctic feeding grounds spanning the Ross Sea across to the Amundsen/Bellingshausen Sea region and the first whale reached the ice-edge around 19 November. The fluke photographs have all been quality controlled and entered into a matching database (Flukematcher). In total there are 133 whales in the Kermadec Islands catalogue; 3 prior to 2015 and 130 from 2015 (129 from our voyage and 1 from the land-based DOC team at Raoul Island). Systematic matching to catalogues from Oceania, east Australia and Antarctica is underway and preliminary analysis found whales matched to the New Caledonia and Niue breeding grounds. We anticipate that other breeding ground origins will be identified once we have completed the systematic matching process. DNA from all 85 samples has been extracted and haplotype assignment to breeding grounds and sex PCR are underway. A sub-sample of tissue (equal numbers of males and females) will be analysed for patterns in isotope values allowing diet assessment.

Overall the early stages of this project are promising. We are on track to understand the importance of Antarctic feeding grounds for the endangered humpback whales passing through New Zealand's northernmost waters.

Other research relevant or specifically linked to the projects above, are listed in Table 18.9.

Table 18.9: Other research linked to MPI Ross Sea Antarctic biodiversity programme.

MPI	ANT2011-01 Stock modelling, fishery effects and ecosystems of the Ross Sea. IPY2007-01 and 02 NZ IPY CAML projects are now complete.
CRI Marine Platform, MBIE	C01X1001 Protecting Ross Sea Ecosystems. Comparative distribution and ecology of *Macrourus caml* and *M. whitsoni* in the Ross Sea region; feeding relationships of fish species in the Ross Sea region; Spatial processes, including spatial marine protection; Ecosystem modelling of the Ross Sea region (Pinkerton et al. 2012, Murphy et al. 2012).
OTHER	Universities NIWA; Lincoln, Canterbury, Otago, Auckland, Waikato; Marsden, Cummings and Lohrer, effects of ocean acidification and warming on under-ice algal productivity and nutrient uptake in coastal Ross Sea habitats.
EMERGING ISSUES	Coastal research and functional ecology-ongoing need. Taxonomic issues for fish and invertebrates (from IPY)ANT 2005-02. Water samples from throughout water column to assess microbial content (from IPY).

Given that the MPI Biodiversity Programme has been running for more than 12 years, and that a number of new strategic documents and directions are emerging across government, it is time to look both back and forward and review the programme to ensure its alignment with more recent strategic documents.

In 2000, five strategic outcomes were built into the MPI (formerly MFish) Biodiversity Research Programme.

That by 2010:

- i. *the MPI 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. *MPI 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 MPI 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 on the first four and part of the fifth of these five outcomes. A missing element is some measure of *'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 agreed objectives include ecosystem-scale studies in the New Zealand marine environment, the classification and characterisation of the biodiversity of near-shore and offshore marine habitats, the role of biodiversity in the functional ecology of marine communities, connectivity and genetic marine biodiversity, the assessment of the effects of climate change and increased ocean acidification, identification of indicators of biodiversity that can be used to monitor change, 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 and offshore fishery-scale trophic studies. The effects of ocean climate change (temperature, acidification) are being 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 to past changes and human impact on New Zealand's marine environment.

A total of 64 projects were commissioned and managed within this 14-year period, yielding over 100 final research reports, most of which have been published through MPI Publications (Marine Biosecurity and Biodiversity Reports and Aquatic Environment and Biodiversity Reports), books, Identification Guides and mainstream scientific literature. A number of other publications are still in preparation. In addition, several workshops have been run through the Programme, including 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, 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 10-year period. For some, the projects have provided critical synergies with MBIE-funded OBIs or projects, while 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 the living conditions and impact 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 very 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 MPI and with other stakeholders (NGOs, industry, other government departments), the Biodiversity Projects have contributed to discussions about Marine Stewardship Council (MSC) certification, to decision papers on aspects of Antarctic management under CAMLR, fulfilling MPI commitments to the New Zealand Biodiversity Strategy, and to MPI progress towards recognising the role of the ecosystem in underpinning sustainable and healthy fisheries production. There are many other examples, e.g., the Programme has contributed towards DOC and MPI 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 MPI Aquatic Environment research programme, the National Aquatic Biodiversity Information System (NABIS), an MPI web-based interactive data access and mapping tool, and the MPI 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 (MSC 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 MPI and the research it has commissioned both across 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 Fish Collection and NIWA National Invertebrate Collection, and Herbarium (macroalgae). All data are held in databases either at MPI, 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 will also incorporate data access from Chatham Challenger and IPY projects. Data from a number of MPI 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, SeaFIC, MfE, Cawthron Institute, NIWA, GNS, LINZ, MAFBNZ, Te Papa, University of Auckland, AUT, University of Otago, MoRST, MFAT, Regional Councils and others. Research planning for 2018–19 and beyond will include 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 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. The Programme has also had close links with Research Data Management and the Observer Programme for certain projects (e.g., trophic studies on the Chatham Rise, ZBD2004-02). With the former restructure of the Ministry of Fisheries and the merger with MAF, and the move to and Fisheries Plans, it important that the Programme develops strong relationships within MPI.

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 across government and among research providers have resulted in many mutually beneficial data and specimen collection, surveys of New Zealand marine biodiversity in New Zealand Territorial Seas, the EEZ and the Ross Sea, groundbreaking 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 (MPI-LINZ, first New Zealand survey of biodiversity in the Ross Sea); Chatham-Challenger (LINZ-MPI-NIWA-DOC first Ocean Survey 20/20 project); NZ IPY-CAML (MPI-LINZ-NIWA; with international and New Zealand-wide collaboration) survey of the Ross Sea as part of International Polar Year; and biodiversity of seamounts (MPI-NIWA-LINZ-MBIE voyages to the Kermadec Arc and on the Chatham Rise). These projects have generated huge geo-referenced datasets and thousands of specimens for Te Papa and National Invertebrate Collections. 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, Kaipara and Manukau Harbours), and along the west coast of the 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 MPI [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 while others were on a regional scale. Recent projects have addressed patterns of marine biodiversity in relation to environmental drivers with ecosystem function. This enabled modelling to predict the distribution of biodiversity in unsurveyed areas of ocean, and evaluation of the vulnerability of biodiversity to perturbations such as climate change, as well as the modelling of 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 MPI [MFish] 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), the need by MfE to report on the marine environment at a national scale (plankton recording programme, Marine Environmental Monitoring Programme). MPI 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 substantially increased awareness within MPI and across 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. Statements of intent and long-term strategic documents such as Fisheries 2030 and Fish Plans have had biodiversity protection and an ecosystem approach to fisheries management objectives explicitly stated. Future research questions will also need to address follow-up of management decisions to assess whether and to what extent the objectives have been achieved.

In 2000, the concept of research on marine biodiversity was hotly debated among stakeholders and the benefit of the research (other than to scientists) was not widely accepted. In 2010, it is clear that much of the research in this Biodiversity Programme has been about defining and mapping the biological diversity of the sea, its roles in marine ecosystem function, threats to these roles and how best biodiversity and its successful protection can be measured. Huge advances have been made in providing new identification tools for major groups (e.g., Coralline algae). Much progress has been made, and the Programme has successfully raised the profile of biodiversity in coastal and ocean environmental management, in particular fisheries management, and biodiversity research uptake into policy and management decisions within MPI and across government.

18.4 CONCLUDING REMARKS

New Zealand is moving into an era of unprecedented and increasing interest in the utilisation of marine resources (Business Growth Agenda 2015). Mineral, petroleum and gas resources are estimated to be worth billions of dollars to the economy (Glasby & Wright 1990), and new environmental legislation has been enacted (the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012). Changes inshore are also taking effect with the Environmental Protection Authority Act passed by Parliament on 11 May 2011. This Act establishes a new Environmental Protection Authority (EPA) as a standalone crown agent from 1 July 2011, which has responsibility for regulating activities under the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012. A new Environmental Reporting Act 2015 has been launched, and options for Marine Protection are underway.

The government has also set national policy direction under the Resource Management Act 1991 to guide decisions affecting freshwater and coastal environments (National Policy Statement for Freshwater Management 2014,[64] New Zealand Coastal Policy Statement 2010).[65]

New Zealand is a signatory to the CBD Aichi-Nagoya Agreement with a new International Decade for Biodiversity that runs 2011–20, and New Zealand's contribution to the identification of EBSAs in the South-west Pacific, and to GOBI. Progress in our knowledge of the marine biodiversity and ecosystem services provided by the marine environment has clearly been made over the last decade. However, we need a more coordinated approach

[64] MfE. About the National Policy Statement for Freshwater Management. Retrieved from http://www.mfe.govt.nz/fresh-water/national-policy-statement/about-nps.

[65] DOC. New Zealand Coastal Policy Statement 2010. Retrieved from http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/coastal-management/nz-coastal-policy-statement-2010.pdf.

across government to link science to policy needs. Essentially we need to know three things; what is out there in the marine environment to use, protect, or manage; how does the ecosystem function; and what are the impacts of natural- and human-induced changes, and what tools will allow for effective monitoring and management of environmental impacts? 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, 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 too soon for evasive action to be taken.

Consumer driven pressures and social awareness of human impacts on the environment, including marine biodiversity, has been increasing and New Zealand's newly launched science challenge 'Sustainable Seas' takes a conventional ecological approach integrated with a social science approach to ecosystem-based management.

CURRENT NZ/EU PARTNERSHIPS:

- BAYESIANMETAFLATS – Spatial organisation of species distributions: hierarchical and scale-dependent patterns and processes in coastal seascapes http://www.kg.eurocean.org/proj.jsp?load=100553.
- Chess – Biogeography of Deep-Water Chemosynthetic Ecosystems http://www.kg.eurocean.org/proj.jsp?load=103314.
- INDEEP – International Network for Scientific Investigation of Deepsea Ecosystems http://www.kg.eurocean.org/proj.jsp?load=103321.
- PHARMASEA – Increasing Value and Flow in the Marine Biodiscovery Pipeline http://www.kg.eurocean.org/proj.jsp?load=100504.
- MAREFRAME – Co-creating Ecosystem-based Fisheries Management Solutions http://www.kg.eurocean.org/proj.jsp?load=500.
- BENTHIS – Studies the impacts of fishing on benthic ecosystems and will provide the science base to assess the impact of current fishing practices http://www.benthis.eu/en/benthis.htm.

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18.6 APPENDIX

TECHNICAL RATIONALE FOR THE GOALS AND TARGETS OF THE STRATEGIC PLAN FOR THE PERIOD 2011–20. UNEP/CBD/COP/10/9 18 JULY 2010.

Strategic goal A: Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society

Strategic actions should be initiated immediately to address, over a longer term, the underlying causes of biodiversity loss. This requires policy coherence and the integration of biodiversity into all national development policies and strategies and economic sectors and at all levels of government. Approaches to achieve this include communication, education and public awareness, appropriate pricing and incentives, and the broader use of planning tools such as strategic environmental assessment. Stakeholders across all sectors of government, society and the economy, including business, will need to be engaged as partners to implement these actions. Consumers and citizens must also be mobilised to contribute to biodiversity conservation and sustainable use, to reduce their ecological footprints and to support action by governments.

[Note: Targets 1–5 not given here.] Targets 6–11 are directly quoted from the document.

Target 6: By 2020, overfishing is ended, destructive fishing practices are eliminated, and all fisheries are managed sustainably.] Or [By 2020, all exploited fish stocks and other living marine and aquatic resources are harvested sustainably [and restored], and the impact of fisheries on threatened species and vulnerable ecosystems are within safe ecological limits.
Overexploitation is the main pressure on marine fisheries globally and the World Bank estimates that overexploitation represents a lost profitability of some $50 billion per year and puts at risk some 27 million jobs and the well-being of more than one billion people. Better fisheries management, which may include a reduction in fishing effort is needed to reduce pressure on ecosystems and to ensure the sustainable use of fish stocks. The specific target should be regarded as a step towards ensuring that all fisheries are sustainable while building upon existing initiatives such as the Code of Conduct for Responsible Fishing. Indicators to measure progress towards this target include the Marine Trophic Index, the proportion of products derived from sustainable sources and trends in abundance and distribution of selected species. Other possible indicators include the proportion of collapsed species, fisheries catch, catch per unit effort, and the proportion of stocks overexploited. Baseline information for several of these indicators is available from the Food and Agriculture Organization of the United Nations.

Target 7: By 2020, areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.
The increasing demand for food, fibre and fuel will lead to increasing losses of biodiversity and ecosystem services if management systems do not become increasingly sustainable with regard to the biodiversity. Criteria for sustainable forest management have been adopted by the forest sector and there are many efforts by governments, indigenous and local communities, NGOs and the private sector to promote good agricultural, aquaculture and forestry practices. The application of the ecosystem approach would also assist with the implementation of this target. While, as yet, there are no universally agreed sustainability criteria, given the diversity of production systems and environmental conditions, each sector and many initiatives have developed their own criteria, which could be used pending the development of a more common approach. Similarly, the use of certification and labelling systems or standards could be promoted as part of this target. Relevant indicators for this target include the area of forest, agricultural and aquaculture ecosystems under sustainable management, the proportion of products derived from sustainable sources and trends in genetic diversity of domesticated animals, cultivated plants and fish species of major socioeconomic importance. Existing sustainability certification schemes could provide baseline information for some ecosystems and sectors. UNEP/CBD/COP/10/9 Page 5 /...

Target 8: By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.
Pollution, including nutrient loading is a major and increasing cause of biodiversity loss and ecosystem dysfunction, particularly in wetland, coastal, marine and dryland areas. Humans have already more than doubled the amount of 'reactive nitrogen' in the biosphere, and business-as-usual trends would suggest a further increase of the same magnitude by 2050. The better control of

sources of pollution, including efficiency in fertiliser use and the better management of animal wastes, coupled with the use of wetlands as natural water treatment plants where appropriate, can be used to bring nutrient levels below levels that are critical for ecosystem functioning, without curtailing the application of fertiliser in areas where it is necessary to meet soil fertility and food security needs. Similarly, the development and application of national water quality guidelines could help to limit pollution and excess nutrients from entering freshwater and marine ecosystems. Relevant indicators include nitrogen deposition and water quality in freshwater ecosystems. Other possible indicators could be the ecological footprint and related concepts, total nutrient use, nutrient loading in freshwater and marine environments, and the incidence of hypoxic zones and algal blooms. Data that could provide baseline information already exist for several of these indicators, including the global aerial deposition of reactive nitrogen and the incidence of marine dead zones (an example of human-induced ecosystem failure).

Target 9: By 2020, invasive alien species are identified, prioritised and controlled or eradicated and measures are in place to control pathways for the introduction and establishment of invasive alien species.

Invasive alien species are a major threat to biodiversity and ecosystem services, and increasing trade and travel means that this threat is likely to increase unless additional action is taken. Pathways for the introduction of invasive alien species can be managed through improved border controls and quarantine, including through better coordination with national and regional bodies responsible for plant and animal health. While well-developed and, globally applicable indicators are lacking, some basic methodologies do exist which can serve as a starting point for further monitoring or provide baseline information. Process indicators for this target could include the number of countries with national invasive species policies, strategies and action plans and the number of countries that have ratified international agreements and standards related to the prevention and control of invasive alien species. One outcome-oriented indicator is trends in invasive alien species while other possible indicators could include the status of alien species invasion, and the Red List Index for impacts of invasive alien species.

Target 10: By 2020, to have minimised the multiple pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification, so as to maintain their integrity and functioning.

Given the ecological inertias related to climate change and ocean acidification, it is important to urgently reduce other pressures on vulnerable ecosystems such as coral reefs so as to give vulnerable ecosystems time to cope with the pressures caused by climate change. This can be accomplished by addressing those pressures that are most amenable to rapid positive changes and would include activities such as reducing pollution and overexploitation and harvesting practices which have negative consequences on ecosystems. Indicators for this target include the extent of biomes ecosystems and habitats (% live coral, and coral bleaching), Marine Trophic Index, the incidence of human-induced ecosystem failure, and the health and well-being of communities who depend directly on local ecosystem goods and services, proportion of products derived from sustainable sources. UNEP/CBD/COP/10/9 Page 6 /...

Strategic goal C: To improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity

Whilst longer-term actions to reduce the underlying causes of biodiversity loss are taking effect, immediate actions, such as protected areas, species recovery programmes, land-use planning approaches, the restoration of degraded ecosystems and other targeted conservation interventions can help conserve biodiversity and critical ecosystems. These might focus on culturally valued species and key ecosystem services, particularly those of importance to the poor, as well as on threatened species. For example, carefully sited protected areas could prevent the extinction of threatened species by protecting their habitats, allowing for future recovery.

Target 11: By 2020, at least [15%][20%] of terrestrial, inland-water and [X%] of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through comprehensive, ecologically representative and well-connected systems of effectively managed protected areas and other means, and integrated into the wider land- and seascape.

Currently, some 13% of terrestrial areas and 5% of coastal areas are protected, while very little of the open oceans are protected. Therefore reaching the proposed target implies a modest increase in terrestrial protected areas globally, with an increased focus on representativity and management effectiveness, together with major efforts to expand marine protected areas. Protected areas should be integrated into the wider land- and seascape, bearing in mind the importance of complementarity and spatial configuration. In doing so, the ecosystem approach should be applied taking into account ecological connectivity and the concept of ecological networks, including connectivity for migratory species. Protected areas should also be established and managed in close collaboration with, and through participatory and equitable processes that recognize and respect the rights of indigenous and local communities, and vulnerable populations. Other means of protection may also include restrictions on activities that impact on biodiversity, which would allow for the safeguarding of sites in areas beyond national jurisdiction in a manner consistent with the jurisdictional scope of the Convention as contained in Article 4. Relevant indicators to measure progress towards this target are the coverage of sites of biodiversity significance covered by protected areas and the connectivity/fragmentation of ecosystems. Other possible indicators include the overlay of protected areas with ecoregions, and the governance and management effectiveness of protected areas. Good baseline information already exists from sources such as the World Database of Protected Areas the Alliance for Zero Extinction, and the IUCN Red List of Threatened Species and the IUCN World Commission on Protected Areas.

19 APPENDICES

19.1 TERMS OF REFERENCE FOR THE AQUATIC ENVIRONMENT WORKING GROUP (AEWG) FOR 2017 ONWARDS

OVERALL PURPOSE

The purpose of the AEWG is to assess, based on scientific information, the effects of (and risks posed by) fishing (for all fisheries in which New Zealand engages), aquaculture, and enhancement on the aquatic environment including:

- 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;
- changes to ecosystem structure and function from fishing, including trophic effects; and
- effects of aquaculture and fishery enhancement on the environment and on fishing.

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 MPI fisheries managers and the Minister responsible for fisheries).

MPI 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 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, aquaculture or enhancement.

PREPARATORY TASKS

1. Prior to the beginning of AEWG meetings each year, MPI 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, MPI-Fisheries or aquaculture staff will alert the relevant AEWG chair prior to the required meetings 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, aquaculture or enhancement impacts, including risks of impacts, and the relative or absolute sensitivity or susceptibility of potentially affected species, populations, habitats, and systems.
4. To estimate and derive appropriate reference points for determining population, system, or environmental status, noting any relevant draft or published management policies (e.g., National Plan of Action or Threat Management Plan). 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.
5. 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.
6. 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.

7. 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 management policies (e.g., National Plan of Action or Threat Management Plan).
8. For species, populations, habitats, or systems for which new assessments are not conducted in the current year, to review and update any existing Fisheries Assessment Plenary report text in order to determine whether the latest reported status summary is still relevant; else to revise the evaluations based on new data or analyses, or other relevant information.
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 ANNUAL 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, aquaculture or enhancement. 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 Principal Advisor Fisheries Science and Aquatic Environment Team 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 a substantial implications for a population, system, or environment or identify risks associated with fishing activity, aquaculture or enhancement; and
    - Consistent performance outside accepted reference points or goals as defined by relevant draft or published management policies (e.g., National Plan of Action or Threat Management Plan).

AEWG MEMBERSHIP 2017

CONVENORS: Rich Ford, Nathan Walker

Membership lists those that have attended at least one meeting in the 2017 calendar year.

MEMBERS: Ed Abraham, Owen Anderson, Karen Baird, Suze Baird, Stephen Beatson, Tiffany Bock, Laura Boren, Christine Bowden, Erin Breen, Susan Chalmers, Tom Clark, Katie Clemens-Seely, Deanna Clement, Richard Cuthbert, Igor Debski, Charlie Edwards, Jack Fenaughty, Dave Foster, Malcolm Francis, Dave Goad, Katrina Goddard, Judi Hewitt, Kristina Hillock, Freydis Hjorvarsdottis, Lyndsey Holland, Sunkita Howard, Krista Hupman, Rosie Hurst, James Jolly, Emma Jones, Jo Lambie, Amanda Leathers, Mary Livingston, Greg Lydon, Lucy Manning, Jennifer Matthews, Andy Mckay, Darryl McKenzie, David Middleton, Rikki Mules, Conor Nielson, Jenny Oliver, Tracey Osborne, Enrique Pardo, Heiko Phillipi, Kris Ramm, Amanda Richards, Yvan Richard, Jim Roberts, Bruce Robertson, Marie-Julie Roux, Paul Scofield, Ben Sharp, Liz Slooten, Geoff Tingley, Rob Tinkler, Ian Tuck, Anton Van Heldon, D'Arcy Webber, Barry Weeber, Jody Weir, Richard Wells, Sam Whinam.

19.2 TERMS OF REFERENCE FOR THE BIODIVERSITY RESEARCH ADVISORY GROUP (BRAG) FOR 2017 ONWARDS

OVERALL PURPOSE

Since 2000, the objectives of the Biodiversity Research Programme have been drawn directly from MFish commitments to Theme 3 of the New Zealand Biodiversity Strategy. Within this framework, the Biodiversity Medium Term Research Plan has been adapted 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 activity.

Within the constraints of the overall purpose of the Programme,

> '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'[1]

and the NZBS definition of biodiversity (the variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystem) 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;
- 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.

MPI also convenes the Aquatic Environment Working Group (AEWG; see above), which has a similar review function to the 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 other MPI sectors 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 the MPI Fisheries Management Group and the Minister responsible for fisheries).

PREPARATORY TASKS

1. Prior to the beginning of BRAG meetings each year, MPI 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.

[1] See MFish Biodiversity Research Programme 2010: Part 1. Context and Purpose.

2. The Ministry's research planning processes should identify most information needs well in advance but, if urgent issues arise, MPI 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. To review, discuss and convey views on the results of marine biodiversity research projects contracted by MPI (formerly Ministry of Fisheries).

It is the responsibility of the BRAG to review, discuss, and convey views on the results of marine biodiversity research projects contracted by MPI and the former Ministry of Fisheries. 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 the BRAG to identify gaps in our knowledge and for developing research plans to address these gaps.

4. Discuss, evaluate, make recommendations and convey views on a 3- to 5-year Medium Term Research Plan.

It is the responsibility of BRAG participants to discuss, evaluate, make recommendations and convey views on a 3- to 5-year Medium Term Research Plan for its particular research area as required. Individual related projects on a species or fishery or research topic need to be integrated into Medium Term Research Plans. The Medium Term Research Plans should encompass research needs and directions for at least the next 3 to 5 years.

The Biodiversity Medium Term Research Plan is aligned to relevant strategic and policy directions such as the 'MPI Statement of Intent' and any Strategic Research Plan (Fisheries 2030, Deepwater 10-year research plan) and fisheries plans developed for the appropriate species/fishery or research area, including biodiversity.

The recommendations on project proposals for the next financial year will be submitted via the Chair of BRAG to the Principal Science Advisor Fisheries (MAF).

5. The Biodiversity Research Programme includes research in New Zealand's TS, EEZ, Extended Continental Shelf, the South Pacific Region and the Ross Sea region and has seven scientific work streams as follows:
   i. To develop ecosystem-scale understanding of biodiversity in the New Zealand marine environment.
   ii. To classify and characterise the biodiversity, including the description and documentation of biota, associated with nearshore and offshore marine habitats in New Zealand.
   iii. To investigate the role of biodiversity in the functional ecology of nearshore and offshore marine communities.
   iv. To assess developments in all aspects of biodiversity, including genetic marine biodiversity and identify key topics for research.
   v. To determine the effects of climate change and increased ocean acidification on marine biodiversity, as well as effects of incursions of non-indigenous species, and other threats and impacts.
   vi. To develop appropriate diversity metrics and other indicators of biodiversity that can be used to monitor change.
   vii. To identify threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation.

BRAG INPUT TO MPI 'AQUATIC ENVIRONMENT AND BIODIVERSITY ANNUAL REVIEW'

6. 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 Fishery Assessment Working Groups.
7. 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 by MPI, the Biodiversity Strategy and international obligations.
8. 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.

9. To advise the Principal Science Advisor Fisheries (MPI) 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;
- any new studies or data that impact on international obligations.

BRAG ATTENDANCE 2016–17

CONVENOR: Mary Livingston

MEMBERS: Sara Bury, David Bowden, Malcolm Clark, Vonda Cummings, Roberta D'Archino, Moira Decima, Britt Graham, Barb Hayden, Cliff Law, Daniel Leduc, Carolyn Lundquist, Alison McDiarmid, Sara Mikaloff-Fletcher, Sadie Mills, Wendy Nelson, Scott Nodder, Darren Parsons, Jim Roberts, Di Tracey, Benton Twist, Ashley Rowden, Matt Pinkerton (NIWA), Rochelle Constantine (University of Auckland), Shane Geange, Debbie Freeman (DOC), Amanda Leathers (WWF), Colin Johnston (AQNZ), Richard Wells (Independent), Sharleen Gargiulo (Deepwater Group), Rich Ford, Malindi Gammon, Lyndsey Holland, Greg Lydon Jennifer Matthews, Tiffany Bock (MPI), Jonathan Gardner (VUW).

19.3 TERMS OF REFERENCE FOR THE ANTARCTIC WORKING GROUP (ANTWG) FOR 2017 ONWARDS

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, fishery 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 in order 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. 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

12. 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.

ANTWG ATTENDANCE 2016–17

CONVENOR: Marine Pomarède

MEMBERS: Anne Boulet, David Bowden, Malcolm Clark, Côme Denechaud, Stuart Hanchet, Kath Large, Craig Marsh, Sophie Mormede, Richard O'Driscoll, Steve Parker, Matt Pinkerton, Jim Roberts, Ashley Rowden, Darren Stevens, Mike Williams (all NIWA), Debbie Freeman (DOC), Barry Weeber (ECO), Brodie Plum, Andy Smith (Talleys), Darryn Shaw (Sanford), Regina Eisert, Toni Wi (University of Canterbury), Jack Fenaughty (Silvifish Resources Ltd), Alistair Dunn, Annie Galland, Bethany Hinton, Conor Nielson, Ben Sharp, Kalolaine Vaipuna (all MPI), Bob Zuur.

19.4 GENERIC TERMS OF REFERENCE FOR FISHERIES ASSESSMENT WORKING GROUPS (FAWGS) FOR 2017 ONWARDS

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 MPI 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), MPI 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, MPI fisheries managers will alert MPI science managers and the Principal Advisor Fisheries Science to unscheduled special cases for which assessments or evaluations are urgently needed.

TECHNICAL OBJECTIVES

3. To review any 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 estimate appropriate MSY-compatible reference points [2] for selected fish stocks for use as reference points for determining stock status, based on the Harvest Strategy Standard for New Zealand Fisheries[3] (the Harvest Strategy Standard).
5. To conduct stock assessments or evaluations for selected fish stocks in order 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 and 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 are not conducted in the current year, to review the existing Fisheries Assessment Plenary report text on the 'Status of the Stocks' in order 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,

[2] 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 the three of these quantities.

[3] Link to the Harvest Strategy Standard: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=208&se=&sd=Asc&filSC=&filAny=False&filSrc=False&filLoaded=False&filDCG=9&filDC=0&filST=&filYr=0&filAutoRun=1.

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 before setting a TAC or TACC.

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 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.
13. To review, and update if necessary, the 'Status of the Stocks' sections of 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 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. 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 Principal Advisor Fisheries Science 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 a base case, (ii) the assessment is the first for a particular stock or the methodology has been substantially altered since the last assessment, and (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.

19.5 MEMBERSHIP AND PROTOCOLS FOR ALL SCIENCE WORKING GROUPS

This document summarises the protocols for membership and participation in all Science Working Groups including Fisheries Assessment Working Groups (FAWGs), the Aquatic Environment Working Group (AEWG), the Biodiversity Research Advisory Group (BRAG), the Highly Migratory Species Working Group (HMS), the South Pacific Working Group (SPACWG), the Antarctic Working Group (ANTWG), and the Marine Amateur Fisheries Working Group (MAFWG).

WORKING GROUP CHAIRS

1. The Ministry will select and appoint the Chairs for Science Working Groups. The Chair will be an MPI fisheries or marine scientist who is an active participant in the Working Group, providing

technical input, rather than simply being a facilitator. Working Group Chairs will be responsible for:

- ensuring that Working Group participants are aware of the Terms of Reference for the Working Group, and that the Terms of Reference are adhered to by all participants;
- setting the rules of engagement, facilitating constructive questioning, and focusing on relevant issues;
- ensuring that all peer review processes are conducted in accordance with the Research and Science Information Standard for New Zealand Fisheries[4] (the Research Standard), and that research and science information is reviewed by the relevant Working Group against the *P R I O R* principles for science information quality (page 6) and the criteria for peer review (pages 12–16) in the Standard;
- requesting and documenting the affiliations of participants at each Working Group meeting that have the potential to be, or to be perceived to be, a conflict of interest of relevance to the research under review (refer to page 15 of the Research Standard). Chairs are responsible for managing conflicts of interest, and ensuring that fisheries management implications do not jeopardise the objectivity of the review or result in biased interpretation of results;
- ensuring that the quality of information that is intended or likely to inform fisheries management decisions, the development of environmental standards or the formulation of relevant fisheries policy is ranked in accordance with the information ranking guidelines in the Research Standard (page 21–23), and that resulting information quality ranks are appropriately documented in the Plenary and the Aquatic Environment and Biodiversity Annual Review (AEBAR);
- striving for consensus while ensuring the transparency and integrity of research analyses, results, conclusions and final reports; and
- reporting on Working Group recommendations, conclusions and action items; and ensuring follow-up and communication with the MPI Principal Advisor Fisheries Science, relevant MPI fisheries management staff, and other key stakeholders.

WORKING GROUP MEMBERS

2. Membership of Science Working groups will be open to any participant with the agreement of the Working Group Chair.
3. Working Groups will consist of the following participants:

    - MPI fisheries science chair – required;
    - research providers – required (may be the primary researcher, or a designated substitute capable of presenting and discussing the agenda item);
    - other scientists not conducting the presented research to act in a peer review capacity;
    - representatives of relevant MPI fisheries management teams; and
    - any interested party who agrees to the standards of participation below.

4. Working Group participants must commit to:

    - participating appropriately in the discussion;
    - resolving issues;
    - following up on agreements and tasks;
    - maintaining confidentiality of Working Group discussions and deliberations (unless otherwise agreed in advance, and subject to the constraints of the Official Information Act);
    - adopting a constructive approach;

[4] Ministry of Fisheries (2011) Research and Science Information Standard for New Zealand Fisheries. Retrieved from https://www.mpi.govt.nz/dmsdocument/3692-research-and-science-information-standard-for-new-zealand-fisheries.

- avoiding repetition of earlier deliberations, particularly where agreement has already been reached;
- facilitating an atmosphere of honesty, openness and trust;
- respecting the role of the Chair; and
- listening to the views of others, and treating them with respect.

5. Participants in Working Group meetings will be expected to declare their sector affiliations and contractual relationships to the research under review, and to declare any substantial conflicts of interest related to any particular issue or scientific conclusion.
6. Working Group participants must adhere to the requirements of independence, impartiality and objectivity listed under the Peer Review Criteria in the Research Standard (pages 12–16). It is understood that Working Group participants will often be representing particular sectors and interest groups, and may be expressing the views of those groups. However, when participating in the review of science information, representatives are expected to step aside from their sector affiliations, and to ensure that individual and sector views do not result in bias in the science information and conclusions.
7. Participants in each Working Group will have access to the corresponding sections of the Science Working Group website including the Working Group papers and other information provided in those sections. Access to Science Working Group websites will generally be restricted to those who have a reasonable expectation of attending at least one meeting of a given Science Working Group each year.
8. Working Group members who do not adhere to the standards of participation (paragraph 4), or who use Working Group papers and related information inappropriately (see paragraph 10), may be requested by the Chair to leave a particular meeting or to refrain from attending one or more future meetings. In more serious instances, members may be removed from the Working Group membership and denied access to the Working Group website for a specified period of time.

WORKING GROUP PAPERS AND RELATED INFORMATION

9. Working Group papers will be posted on the MPI-Fisheries website prior to meetings if they are available. As a general guide, PowerPoint presentations and draft or discussion papers should be available at least two working days before a meeting, and near-final papers should be available at least five working days before a meeting if the Working Group is expected to agree to the paper. However, it is also likely that some papers will be made available for the first time during the meeting due to time constraints. If a paper is not available for sufficient time before the meeting, the Chair may provide for additional time following the meeting for additional comments from Working Group members.

10. Working Group papers are 'works in progress' intended to facilitate the discussion of analyses by the Working Groups. They often contain preliminary results that are receiving peer review for the first time and, as such, may contain errors or preliminary analyses that will be superseded by more rigorous work. For these reasons, no-one may release the papers or any information contained in these papers to external parties. In general, Working Group papers should not be cited. Exceptions may be made in rare instances by obtaining permission in writing from the Principal Advisor Fisheries Science, and the authors of the paper. It is also anticipated that Working Group participants who are representing others at a particular Working Group meeting or series of such meetings may wish to communicate preliminary results to the people they are representing. Participants, along with recipients of the information, are required to exercise discretion in doing this, and to guard against preliminary results being made public.

11. From time to time, MPI commissions external reviews of analyses, models or issues. Terms of Reference for these reviews and the names of external reviewers may be provided to the Working Group for information or feedback. It is extremely important to the proper conduct of these reviews that all contact with the reviewers is through the Chair of the Working Group or the Principal Advisor Fisheries Science. Under no

circumstances should Working Group members approach reviewers directly until after the final report of the review has been published.

WORKING GROUP MEETINGS

12. Meetings will take place as required, generally January–April and July–November for FAWGs and throughout the year for other Working Groups (AEWG, BRAG, HMSWG, SPACWG, ANTWG and MAFWG).
13. A quorum will be reached when the Chair, the designated presenter, and at least three other technical experts are present. In the absence of a quorum, the Chair may decide to proceed as a sub-group, with outcomes being discussed with the wider Working group via email or taken forward to the next meeting at which a quorum is formed.
14. The Chair is responsible for deciding, with input from the entire Working Group, but focusing primarily on the technical discussion and the views of technical expert members:

    - the quality and acceptability of the information and analyses under review;
    - the way forward to address any deficiencies;
    - the need for any additional analyses;
    - contents of research reports, Working Group reports and AEBAR chapters;
    - choice of best models and sensitivity analyses to be presented; and
    - the status of the stocks, or the status/performance in relation to any relevant environmental standards or targets.

15. The Chair is responsible for facilitating a consultative and collaborative discussion.
15. Working Group meetings will be run formally, with agendas pre-circulated, and formal records kept of recommendations, conclusions and action items.
16. A record of recommendations, conclusions and action items will be posted on the MPI-Fisheries website after each meeting has taken place.
17. Data upon which analyses presented to the Working Groups are based must be provided to MPI in the appropriate format and level of detail in a timely manner (i.e., the data must be available and accessible to MPI; however, data confidentiality concerns mean that some data may not necessarily be made available to Working Group members).
18. Working Group processes will be evaluated periodically, with a view to identifying opportunities for improvement. Terms of Reference and the Membership and Protocols may be updated as part of this review.
19. MPI fisheries scientists and science officers will provide administrative support to the Working Groups.

INFORMATION QUALITY RANKING

20. Science Working Groups are required to rank the quality of research and science information that is intended or likely to inform fisheries management decisions, in accordance with the science information quality ranking guidelines in the Research Standard (pages 21–23). Information quality rankings should be documented in Working Group reports and, where appropriate, in Status of Stock summary tables. Note that:

    - Working Groups are not required to rank all research projects and analyses, but key pieces of information that are expected or likely to inform fisheries management decisions, the development of environmental decisions or the formulation of relevant policy should receive a quality ranking;
    - explanations substantiating the quality rankings will be included in Working Group reports. In particular, the quality shortcomings and concerns for moderate/mixed and low quality information should be documented; and
    - the Chair, working with participants, will determine which pieces of information require a quality ranking. Not all information resulting from a particular research project would be expected to achieve the same quality rank, and different quality ranks may be assigned to different components, conclusions or pieces of information resulting from a particular piece of research.

RECORD KEEPING

21. The overall responsibility for record-keeping rests with the Chair of the Working Group, and includes:

- keeping notes on recommendations, conclusions and follow-up actions for all Working Group meetings, and to ensure that these are available to all members of the Working Group and the Principal Advisor Fisheries Science in a timely manner. If full agreement on the recommendations or conclusions cannot readily be reached amongst technical experts, then the Chair will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes; and
- compiling a list of generic assessment issues and specific research needs for each stock, species or environmental issue under the purview of the Working Group, for use in subsequent research planning processes.

19.6 GENERIC TERMS OF REFERENCE FOR RESEARCH ADVISORY GROUPS (SEPT 2010)

OVERALL PURPOSE

1. The purpose of the Research Advisory Groups (RAGs) is to develop research proposals to meet management information needs and support standards development.

CONTEXT

2. To assist RAG members with their work this section outlines the wider process that RAGs will operate within.

Fisheries Plans will guide the management of fisheries

3. From 1 July 2011 the Ministry of Fisheries (MFish) will be using Fisheries Plans in the following five areas to guide the management of fisheries:

   - Deepwater
   - Highly Migratory Species
   - Inshore – Finfish
   - Inshore – Freshwater
   - Inshore – Shellfish

4. In each of those five areas there will be:

   - A Fisheries Plan that sets out management objectives over a 5-year period.
   - An Annual Operational Plan that sets out what will be done in a financial year to help meet those objectives, including in the areas of science research, compliance and observer coverage (i.e., the Annual Operational Plan will be where priorities are set each year). Note that external stakeholders will have an opportunity to provide comment on prioritisation through draft Annual Operational Plans.
   - An Annual Review Report that will assess progress made against the management objectives, and help identify gaps to be considered in setting the next set of priorities.

RAGs will largely be aligned to the Fisheries Plan areas

5. There will be a RAG for each of the five Fisheries Plan areas above.
6. In addition there will be a RAG for Aquatic Environment (Standards), for research needed to support standards development, and another for Antarctic research. (Note that biodiversity research is dealt with through a separate process that has more of a cross-agency focus).

RAGs will develop research proposals to be considered as part of a subsequent prioritisation process

7. As part of the process for developing the Annual Operational Plans, the identification and prioritisation of science research will broadly occur as follows:

   i. MFish fisheries managers will identify the fisheries management objectives and information needs that they want the relevant RAG to consider. This will be done in conjunction with MFish scientists, and will draw on the following:

      - The relevant Annual Review Report discussed above;
      - Existing research plans;
      - Science Assessment Working Groups' feedback arising from research that has been evaluated previously;
      - Ad-hoc issues as they arise;

- Initial indications of the available budget.

ii. The RAGs will then develop proposals for scientific research to meet those management and information needs.

iii. MFish fisheries managers will then run a process for prioritising the research proposals that have been developed and updating multi-year research plans, in conjunction with MFish scientists. This will be part of the wider process for developing Annual Operational Plans.

8. In the Aquatic Environment (Standards) and Antarctic areas a similar process will be followed to that above, involving relevant MFish managers.
9. In practice, these processes are likely to iterate between the above steps, e.g., when prioritising research proposals fisheries managers may identify additional questions that they want a RAG to consider.
10. RAGs will only be convened when necessary. If, for example, all of the research for the coming year under review has previously been approved as part of a multi-year funding package for an area, and no additional management needs have emerged, the relevant RAG will not be convened.
11. During 2010–11 RAGs will be used, as required, in all areas except Inshore, given that the three Inshore Fisheries Plans are still being developed through the year. For the Inshore areas a transitional process will be used, with RAGs commencing during 2011–12.

RESEARCH PROPOSALS

12. RAGs will provide recommendations to fisheries managers on research to meet management needs. This section provides more detail on the research proposals that the RAGs will produce.
13. The RAGs will produce an initial set of project proposals to meet the management and information needs provided to the RAG, for consideration in the subsequent prioritisation process.
14. The proposals may be in the form of multi-year projects where appropriate.
15. While the prioritisation of research is outside the scope of the work of the RAGs, the proposals will include information on potential cost and feasibility to guide decisions on prioritisation. Cost estimates should be specified as ranges so as to not unduly influence subsequent research provider costings.
16. Where the RAG identifies more than one desirable option for scientific research to meet management and information needs, the RAG's proposals will cover those options, their relative pros and cons, their respective potential costs, and the RAG's recommendation as to the preferred option.
17. Once prioritisation decisions have been made on the initial set of research proposals, the RAG may be asked to produce more fully developed project proposals for inclusion in the relevant Annual Operational Plan, and for the purposes of cost recovery consultation and tendering.

MEMBERSHIP

18. Membership of RAGs is expertise-based.
19. Membership will be by invitation from MFish only.
20. A RAG will consist of a core group of one MFish scientist and one manager from the relevant Fisheries Plan or Standards team, with the option to 'call in' relevant technical expertise (internal and/or external) as needed.
21. External participants will be paid for their time. This will include preparing for and attending RAG meetings, and any time spent writing proposals.

PROTOCOLS

22. All RAG members will commit to:

    - participating in the discussion in an objective and unbiased manner;
    - resolving issues;
    - following up on agreements and tasks;
    - adopting a constructive approach;
    - facilitating an atmosphere of honesty, openness and trust;
    - having respect for the role of the Chair; and
    - listening to the views of others, and treating them with respect.

23. RAG meetings will be run formally with agendas pre-circulated and formal records kept of recommendations, conclusions and action items.
24. Participants who do not adhere to the standards of participation may be requested by the Chair to

leave a particular meeting or, in more serious instances, will be excluded from the RAG.

Chairpersons

25. The Chair of each RAG will be an MPI scientist with appropriate expertise.
26. The Chair commits to undertaking the following roles:

- The Chair is an active participant in RAGs, who also provides technical input, rather than simply being a facilitator.
- The Chair is responsible for: setting the rules of engagement; promoting full participation by all members; facilitating constructive questioning; focusing on relevant issues; reporting on RAG recommendations, conclusions and action items, and ensuring follow-up; and communicating with relevant MFish managers.

27. The Chair is responsible for facilitating consultative and collaborative discussions.

Decision-making

28. The Chair is responsible for working towards an agreed view of the RAG members on their recommendations to the fisheries manager, but where that proves not to be possible then the Chair is responsible for determining the final recommendation. Minority views should be clearly represented in proposals in those cases.
29. A record of recommendations, conclusions and action items will be circulated by e-mail after each meeting by the Chair.
30. Each RAG round will be evaluated by MFish, with a view to identifying opportunities to improve the process. The Terms of Reference may be updated as part of this review.

Non-disclosure agreements

31. Participants may be asked to sign a Non-Disclosure Agreement relating to documents that disclose cost details.

Conflicts of interest

32. New Zealand is a small country and fisheries research is a relatively limited market, even internationally. People with the necessary skills and knowledge to participate in this advisory process may also have close working relationships with industry, research providers and other stakeholders. This will apply to nearly all external members of a RAG.
33. Participants will be asked to declare any 'actual, perceived or likely conflicts of interest' before involvement in a RAG is approved, and any new conflicts that arise during the process should be declared immediately. These will be clearly documented by the Chair.
34. Management of conflicts of interest will be determined by the Chair in consultation with Fisheries Managers, and approved by the Deputy Chief Executive, Fisheries Management prior to meetings commencing.

Frequency of meetings

Relevant MFish managers, in consultation with the Chair of the RAG, will decide on the frequency and timing of RAG meetings.

Documents and record-keeping

23. Unless signalled by the Chair, all RAG documents (papers, agendas, formal records of recommendations, conclusions and action items) will be available to all interested parties through the Ministry of fisheries website (www.fish.govt.nz), except where confidentiality is required for reasons of commercial sensitivity (e.g., cost estimates).
24. RAG documents will be distributed securely.
25. Participants who use RAG papers inappropriately may not be invited to subsequent RAG meetings.
26. The overall responsibility for record-keeping rests with the Chair and includes:

- Records of recommendations, conclusions and follow-up actions for all RAG meetings and to ensure that these are available in a timely manner.
- If full agreement on the recommendations or conclusions cannot readily be reached amongst technical experts, then the Chair will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

19.7 FISHERIES 2030

USE OUTCOME – Fisheries resources are used in a manner that provides the greatest overall economic, social, and cultural benefit. This means having:

- An internationally competitive and profitable seafood industry that makes a significant contribution to our economy.
- High-quality amateur fisheries that contribute to the social, cultural, and economic well-being of all New Zealanders.
- Thriving customary fisheries, managed in accordance with kaitiakitanga, supporting the cultural well-being of iwi and hapū.
- Healthy fisheries resources in their aquatic environment that reflect and provide for intrinsic and amenity value.

GOVERNANCE CONDITIONS – Fundamental to achieving our goal is the recognition that our approach must be based on sound governance. This means having arrangements that lead to:

- The Treaty partnership being realised through the Crown and Māori clearly defining their respective rights and responsibilities in terms of governance and management of fisheries resources.
- The public having confidence and trust in the effectiveness and integrity of the fisheries and aquaculture management regimes.
- All stakeholders having rights and responsibilities related to the use and management of fisheries resources that are understood and for which people can be held individually and collectively accountable.
- Having an enabling framework that allows stakeholders to create optimal economic, social, and cultural value from their rights and interests.
- An accountable, responsive, dynamic, and transparent system of management.

Fisheries 2030 draws on a number of values and principles. These seek to outline the behaviour and approach that should be used to undertake the actions, make decisions, and achieve the goal for New Zealand fisheries.

VALUES

- Tikanga: the Mäori way of doing things; correct procedure, custom, habit, lore, method, manner, rule, way, code, meaning, reason, plan, practice, convention. It is derived from the word tika meaning 'right' or 'correct'.
- Kaitiakitanga: The root word in kaitiakitanga is tiaki, which includes aspects of guardianship, care, and wise management. Kaitiakitanga is the broad notion applied in different situations.
- Kotahitanga: Collective action and unity.
- Manaakitanga: Manaakitanga implies a duty to care for others, in the knowledge that at some time others will care for you. This can also be translated in modern Treaty terms as 'create no further grievances in the settlement of current claims'.
- Integrity: Be honest and straightforward in our dealings with one another. If we agree to do something we will carry it out.
- Respect: Treat each other with courtesy. We will respect each other's right to have different values and hold different opinions.
- Constructive relationship: Strive to build and maintain constructive ways of working with each other, which can endure.
- Achieving results: Focus on producing a solution rather than just discussing the problem.

PRINCIPLES

- Ecosystem-based approach: We apply an ecosystem-based approach to fisheries management decision-making.
- Conserve biodiversity: Use should not compromise the existence of the full range of genetic diversity within and between species.
- Environmental bottom lines: Biological standards define the limits of extraction and impact on the aquatic environment.
- Precautionary approach: Particular care will be taken to ensure environmental sustainability where information is uncertain, unreliable, or inadequate.
- Address externalities: Those accessing resources and space should address the impacts their activities have on the environment and other users.
- Meet Settlement obligations: Act in ways that are consistent with the Treaty of Waitangi principles and deliver settlement obligations.

- Responsible international citizen: Manage in the context of international rights, obligations, and our strategic interests.
- Inter-generational equity: Current use is achieved in a manner that does not unduly compromise the opportunities for future generations.
- Best available information: Decisions need to be based on the best available and credible biological, economic, social, and cultural information from a range of sources.
- Respect rights and interests: Policies should be formulated and implemented to respect established rights and interests.
- Effective management and services: Use least-cost policy tools to achieve objectives where intervention is necessary and ensure services are delivered efficiently.
- Recover management costs for the reasonable expenses of efficiently provided management and services, from those who benefit from use, and those who cause the risk or adverse effect.
- Dynamic efficiency: Frameworks should be established to allow resources to be allocated to those who value them most.

Fisheries 2030 includes a 'plan of action' for the five years from 2009, including: improving the management framework; supporting aquaculture and international objectives; ensuring sustainability of fish stocks; improving fisheries information; building sector leadership and capacity; meeting obligations to Māori; and enabling collective management responsibility. The key components guiding this document are ensuring sustainability of fish stocks and improving fisheries information.

ENSURING SUSTAINABILITY OF FISH STOCKS

- Setting and implementing fisheries harvest strategy standards.
- Setting and monitoring environmental standards, including for threatened and protected species and seabed impacts.
- Enhancing the framework for fisheries management planning, including the use of decision rules to adjust harvest levels over time.

IMPROVING FISHERIES INFORMATION

- Determining best options for information collection on catch from amateur fisheries, including the implementation of charter boat reporting.
- Improving our knowledge of fish stocks and the environmental impacts of fishing through long-term research plans.
- Gaining access to increased research and development funding.

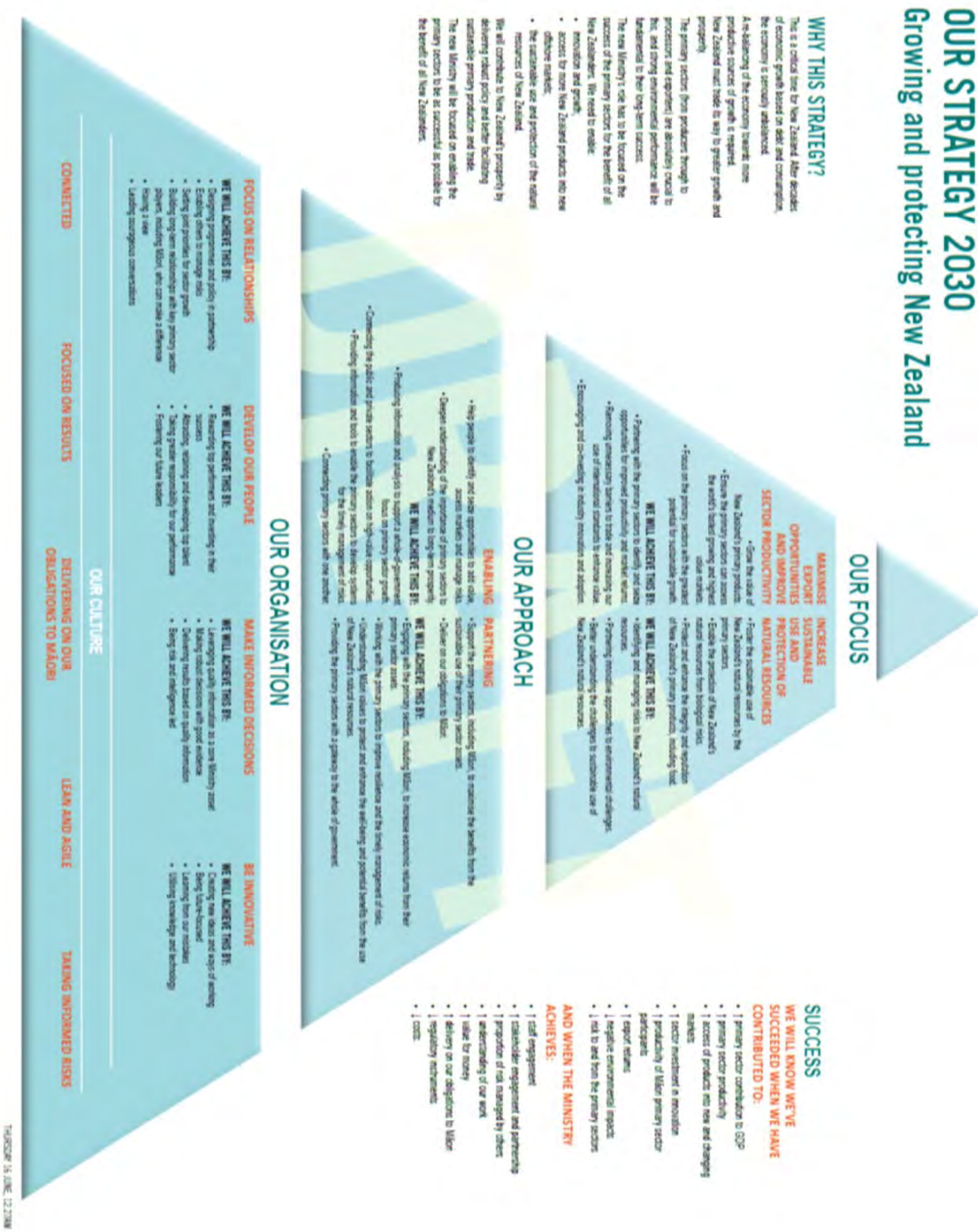

Figure 19.1: Our Strategy 2030: Growing and protecting New Zealand.

19.8 OTHER STRATEGIC POLICY DOCUMENTS

19.8.1 BIODIVERSITY STRATEGY

New Zealand's Biodiversity Strategy was launched in 2000 in response to the decline of New Zealand's indigenous biodiversity — described in the State of New Zealand's Environment report as our 'most pervasive environmental issue'. It can be found on the government's biodiversity website at: http://www.doc.govt.nz/biodiversity.

The Strategy also reflects New Zealand's commitment, through ratification of the international Convention on Biological Diversity, to help stem the loss of biodiversity worldwide. Strategic Priority 7 of the strategy was 'To manage the marine environment to sustain biodiversity'. Fishing practices, the effects of activities on land, and biosecurity threats are identified as constituting the areas of greatest risk to marine biodiversity. Pertinent objectives and summarised actions from the strategy are as follows:

Objective 3.1: Improving our knowledge of coastal and marine ecosystems (Substantially increase our knowledge of coastal and marine ecosystems and the effects of human activities on them, especially assessing the importance of, and threats facing, marine biodiversity, and establishing environmental monitoring capabilities to assess the effectiveness of measures to avoid, remedy or mitigate impacts on marine biodiversity).

Objective 3.4: Sustainable marine resource use practices (Protect biodiversity in coastal and marine waters from the adverse effects of fishing and other coastal and marine resource uses, especially maintaining harvested species at sustainable levels, integrating marine biodiversity protection into an ecosystem approach, applying a precautionary approach, identifying marine species and habitats most sensitive to disturbance, and integrating environmental impact assessments into fisheries management decision making.)

Objective 3.6: Protecting marine habitats and ecosystems (Protect a full range of natural marine habitats and ecosystems to effectively conserve marine biodiversity, using a range of appropriate mechanisms, including legal protection, especially establishing a network of areas that protect marine biodiversity.)

Objective 3.7: Threatened marine and coastal species management (Protect and enhance populations of marine and coastal species threatened with extinction, and prevent additional species and ecological communities from becoming threatened.)

In addition to its annual reviews, the Biodiversity Strategy was reviewed by Green and Clarkson at the end of its 5-year term. This review was published in 2006 (http://www.doc.govt.nz/documents/conservation/nzbs-report.pdf). Most relevant to this synopsis were their findings on Objective 3.4 (Sustainable marine resource use) where they cited 'Moderate progress'. *'The policy move towards adopting a more ecosystem approach to fisheries management should be encouraged and strengthened. We acknowledge, however, the difficulties associated with obtaining the necessary information to make this approach effective. There are links to Objective 3.1 and the need for a more coordinated approach to identifying priority areas for marine research.'*

19.8.2 BIOSECURITY STRATEGY

In its 2003 Biosecurity Strategy, the Ministry of Agriculture and Forestry's Biosecurity NZ defined biosecurity as 'the exclusion, eradication or effective management of risks posed by pests and diseases to the economy, environment and human health'. New Zealand is highly dependent on effective biosecurity measures because our indigenous flora, fauna, biodiversity, and, consequently, our primary production industries, including fisheries are uniquely at risk from invasive species. A complementary Biosecurity Science Strategy for New Zealand was developed in 2007 to address the science expectations of the Biosecurity Strategy. The science strategy identified the need to:

- prioritise science needs;
- minimise biosecurity risks at the earliest stage possible by increasing focus on research that is strategic and proactive;
- improve planning, integration and communication in the delivery of science;
- ensure research outputs can be used effectively to improve biosecurity operations and decision making.

19.8.3 MARINE PROTECTED AREAS POLICY

The Marine Protected Areas (MPA) Policy and Implementation Plan was released for consultation in December 2005 jointly by the Ministry of Fisheries and Department of Conservation. It confirmed Government's commitment to ensuring that New Zealand's marine biodiversity was protected, and established MPA Policy as a key component of that commitment. The MPA Policy objective is to protect marine biodiversity by establishing a network of Marine Protected Areas that is comprehensive and representative of New Zealand's marine habitats and ecosystems. The Policy involved a four-stage approach to implementation:

Stage 1: Development of the approach to classification, formulation of a standard of protection, and mapping of existing protected areas and/or mechanisms. Scientific workshops will be used to assist with the process, and the results will be put on the website for comment.

Stage 2: Development of the MPA inventory, identification of gaps in the MPA network, and prioritisation of new MPAs.

Stage 3: Establishment of new MPAs to meet gaps in the network. This will be undertaken at a regional level and a national process will be followed for offshore MPAs.

Stage 4: Evaluation and monitoring.

Stage 1 and the inventory specified for Stage 2 are complete and regional forums were established for the Subantarctic and West Coast bioregions.

The link for the stage 2 report is at:

http://www.doc.govt.nz/publications/conservation/marine-and-coastal/marine-protected-areas/coastal-marine-habitats-and-marine-protected-areas-in-the-new-zealand-territorial-sea-a-broad-scale-gap-analysis.

In June 2009, these planning forums released consultation documents on implementation of the MPA Policy in their bioregions:

Consultation Document – Implementation of the Marine Protected Areas Policy in the Territorial Seas of the Subantarctic Biogeographic Region of New Zealand:

http://www.marinenz.org.nz/documents/subantarctics-mpa-policy-consultation-document.pdf.

The MPA Classification, Protection Standard, Implementation Guidelines, together with a summary of subsequent consultation processes around implementing the policy can be found at:

http://www.doc.govt.nz/Documents/conservation/marine-and-coastal/marine-protected-areas/mpa-classification-protection-standard.pdf.

19.8.4 REVISED COASTAL POLICY STATEMENT

The revised New Zealand Coastal Policy Statement (NZCPS) came into force in December 2010, replacing the original 1994 NZCPS. The statement is to be applied, as required by the Resource Management Act 1991 (RMA), by persons exercising functions and powers under that Act. The documentation can be read on the Department of Conservation's website at:

http://www.doc.govt.nz/publications/conservation/marine-and-coastal/new-zealand-coastal-policy-statement/new-zealand-coastal-policy-statement-2010.

The NZCPS does not directly apply to fisheries management decision-making, although the Minister of Fisheries is required to have regard to the Statement when making decisions on sustainability measures under s11 of the Fisheries Act. In addition, this synopsis include chapters on land use issues and habitats of particular significance for fisheries management for which the main threats are managed under the RMA (e.g., land-use practices could increase sedimentation and affect the estuarine nursery grounds of important fishstocks). In other areas, management of effects under the RMA can complement management of the effects of fishing (e.g., complementary management of the habitat and bycatch of a protected species). The following objectives and policies are considered relevant (numbering as per NZCPS, text in parentheses summarises subheadings in the Statement of most relevance to fisheries values):

Objective 1: To safeguard the integrity, form, functioning and resilience of the coastal environment and sustain its ecosystems, including marine and intertidal areas, estuaries, dunes and land (especially by maintaining or enhancing natural biological and physical processes in the coastal environment).

Objective 6: To enable people and communities to provide for their social, economic, and cultural wellbeing and their health and safety, through subdivision, use, and development (especially by recognising that the protection of habitats of living marine resources contributes to social, economic and cultural wellbeing and that the potential to utilise coastal marine natural resources should not be compromised by activities on land).

Policy 5: Land or waters managed or held under other Acts (especially to consider effects on coastal areas held or managed under other Acts with conservation or protection purposes and to avoid, remedy or mitigate adverse effects of activities in relation to those purposes).

Policy 8: Aquaculture: Recognise the significant existing and potential contribution of aquaculture to the social, economic and cultural well-being of people and communities (especially by taking account of the social and economic benefits of aquaculture, recognising the need for high water quality, and including provision for aquaculture in the coastal environment).

Policy 11: Indigenous biodiversity: To protect indigenous biological diversity in the coastal environment (especially by avoiding, remedying or mitigating adverse effects on: habitats that are important during the vulnerable life stages of indigenous species; ecosystems and habitats that are particularly vulnerable to modification; and habitats of indigenous species that are important for recreational, commercial, traditional or cultural purposes).

Policy 21: Enhancement of water quality: Where the quality of water in the coastal environment has deteriorated so that it is having a significant adverse effect on ecosystems, natural habitats, or water based recreational activities, or is restricting existing uses, such as aquaculture, shellfish gathering, and cultural activities, give priority to improving that quality.

Policy 22: Sedimentation (especially with respect to impacts on the coastal environment).

Policy 23: Discharge of contaminants (especially with respect to impacts on ecosystems and habitats).

19.8.5 MANAGEMENT OF ACTIVITIES IN THE EEZ

Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012. The Act manages the environmental effects of activities in New Zealand's oceans. The legislation aims to protect our oceans from the potential environmental risks of activities like petroleum exploration activities, seabed mining, marine energy generation and carbon capture developments.

The Resource Management Act regulates natural resource management activities on land and in the Territorial Sea out to 12 nautical miles. Fishing and shipping are also regulated by other Acts. The EEZ Act does not override these other controls that already exist in the EEZ. Beyond 12 nautical miles New Zealand has historically had no means to assess and regulate the environmental effects of many other activities. The EEZ Act fills that regulatory gap and manages the previously unregulated adverse environmental effects of activities in the EEZ and continental shelf. Before the EEZ Act was passed there was a gap in our domestic legislation.

The EEZ Act sets up a framework for managing the effects of activities in the EEZ and continental shelf. The text of the Act can be found on the New Zealand Legislation website.

The EEZ legislation to manage effects other than those caused by fishing do not directly apply to fisheries management decision-making under the Fisheries Act. However, there are issues around the management of cumulative effects (e.g., of more than one activity on benthic communities) and around effects of any proposed new activities in the EEZ on fishing activity already occurring. Some projects already completed or currently underway are likely to be useful for these processes (e.g., detailed maps of fishing effort produced under ENV2001/07 and BEN2006/01 and enhancements of the Marine Environment Classification produced under ZBD2005-02 for demersal fishes and BEN2006/01A for benthic invertebrates).

19.8.6 NATIONAL PLAN OF ACTION TO REDUCE THE INCIDENTAL CATCH OF SEABIRDS IN NEW ZEALAND FISHERIES

New Zealand released its first National Plan of Action (NPOA) to reduce the Incidental Catch of Seabirds in New

Zealand Fisheries in April 2004. That document is available online at:

http://www.doc.govt.nz/documents/conservation/native-animals/birds/npoa.pdf.

A completely revised and refreshed NPOA-Seabirds was released in March 2013. A resources page was added to the MPI (Fisheries) website to provide access to this plan, its supporting risk assessment documents, a web-based reporting system for protected species captures, and information on MPI's fisheries planning processes that will be the vehicle for implementation:

https://fs.fish.govt.nz/Page.aspx?pk=108.

The 2013 NPOA-Seabirds can be found at:

https://www.mpi.govt.nz/dmsdocument/3962-national-plan-of-action-2013-to-reduce-the-incidental-catch-of-seabirds-in-new-zealand-fisheries.

The 2013 NPOA covers all New Zealand fisheries and has a long-term objective that *'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.'*

There are high-level subsidiary objectives related to practical aspects, biological risk, research and development, and international issues.

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.

19.8.7 NEW ZEALAND NATIONAL PLAN OF ACTION FOR THE CONSERVATION AND MANAGEMENT OF SHARKS

The New Zealand National Plan of Action (NPOA) for the Conservation and Management of Sharks (2013) was approved by the Minister of Fisheries on 9 January 2014. The purpose of the NPOA-Sharks is to ensure the conservation and management of sharks and their long-term sustainable use. It also contains a set of actions in order to meet this purpose. The document is available online at:

http://www.mpi.govt.nz/dmsdocument/1138-national-plan-of-action-for-the-conservation-and-management-of-sharks-2013.

19.8.8 NATIONAL SCIENCE CHALLENGES

The National Science Challenges were conceived to tackle some of the biggest science-based issues and opportunities facing New Zealand. They were designed to take a more strategic approach to the government's science investment by targeting a series of goals, which, if achieved, would have major and enduring benefits for New Zealand. The Challenges provide an opportunity to align and focus New Zealand's research on large and complex issues by drawing scientists together from different institutions and across disciplines to achieve a common goal through collaboration.

Many of the issues facing New Zealand require new knowledge obtained through science and research. The government has launched the Challenges to provide a means to address the most pressing of these complex issues. The Challenges will seek answers to questions of national significance to New Zealand by focusing effort and providing additional focus on key areas. The Challenges provide an opportunity to identify which issues are most important to New Zealand and will allow government to take a targeted, cross-government approach to addressing them.

Each Challenge includes both new funding and funds that will become available as current MBIE research contracts mature. Relevant CRI core funding will also be invested in Challenges, where CRIs are part of a Challenge collaboration. The new Challenge money comprises $73.5 million over four years in Budget 2013, in addition to the $60 million allocated in Budget 2012, and $30.5 million per year thereafter.

The eleven research areas identified for National Science Challenge funding (asterisks mark those Challenges potentially relevant to fisheries and the marine environment) were:

1. High Value Nutrition
2. The Deep South *
3. New Zealand's Biological Heritage *
4. Sustainable Seas *
5. A Better Start
6. Resilience to Nature's Challenges *
7. Science for Technological Innovation
8. Ageing Well
9. Healthier Lives
10. Our Land and Water *
11. Building Better Homes, Towns and Cities

See also: http://www.mbie.govt.nz/info-services/science-innovation/national-science-challenges.

The Ministry for Business, Innovation and Employment administers the Challenges and issued Requests for Proposals for four of the Challenges in October 2013 and for the remainder in February 2014. Given that the Challenges represent a radically different approach to research in New Zealand, and required substantial collaboration between science organisations, it is perhaps not surprising that designing and contracting the work has taken some time. The following Challenges of relevance to fisheries and marine systems have been launched (as at December 2014, listed in order of their launch):

The Deep South — Te Kōmata o Te Tonga — was launched on 5 August 2014 with a headline of *Understanding the role of the Antarctic and the Southern Ocean in determining our climate and our future environment*. The mission of this Challenge is to transform the way New Zealanders adapt, manage risk, and thrive in a changing climate. Working with communities and industry we will bring together new research approaches to determine the impacts of a changing climate on our climate-sensitive economic sectors, infrastructure and natural resources to guide planning and policy. This will be underpinned by improved knowledge and observations of climate processes in the Southern Ocean and Antarctica – our Deep South – and will include development of a world-class earth systems model to predict Aotearoa/New Zealand's climate. Further information can be found at: http://www.deepsouthchallenge.co.nz.

New Zealand's Biological Heritage — Ngā Koiora Tuku Iho — was launched on 29 August 2014 with a headline of *Protecting and managing our biodiversity, improving our biosecurity, and enhancing our resilience to harmful organisms*. This Challenge does not consider marine systems as such, but includes estuarine systems and close liaison between this Challenge and Sustainable Seas will be necessary to ensure important biological systems and processes are covered. Further information can be found at: http://www.biologicalheritage.nz.

Sustainable Seas — Ko ngā moana whakauka — was launched on 4 September 2014 with a headline of *Enhance utlilisation of our marine resources within environmental and biological constraints*. The aim of this Challenge is to enhance use of New Zealand's vast marine resources, while ensuring that our marine environment is understood, cared for, and used wisely for the benefit of all, now and in the future. This requires 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 economic development. Thus, this is the Challenge most closely associated with fisheries management. Further information can be found at: http://sustainableseaschallenge.co.nz.

19.9 APPENDIX OF AQUATIC ENVIRONMENT AND BIODIVERSITY FUNDED AND RELATED PROJECTS

The following listing of projects are those relevant to aquatic environment research that have been through research planning and subsequently been funded by the Ministry of Fisheries (MFish), the Ministry for Primary Industries (MPI) or the fishing industry. These projects have been ordered by the research themes:

1. Protected species (PRO)
2. Non-protected bycatch (NPB)
3. Benthic impacts (BEN)
4. Ecosystem effects (ECO)
5. Biodiversity (ZBD)

Within these themes projects are ordered chronologically (from the most recent to the oldest). A list of references cited within the table is included at the end of this appendix.

Each project or row of the table is described by a project number (used by MFish/MPI), a project title, specific objectives (where there are many objectives and some are clearly not relevant to aquatic environment research they may not be listed), project status and any relevant citations from the project.

Citations listed below can be accessed differently depending upon the type of output. Finalised FARs (Fisheries Assessment Reports) and AEBRs (Aquatic Environment and Biodiversity Reports), historical FARDs (Fisheries Assessment Research Documents) and MMBRs (Marine Biodiversity and Biosecurity Reports), and some FRRs (final Research Reports) can be found at: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=209.
Increasingly, reports will be available from the MPI website at: https://www.mpi.govt.nz/news-and-resources/publications. For unpublished documents or those not available on either of these websites please contact Science.Officer@mpi.govt.nz. Every attempt has been made to make this table comprehensive and correct, but if any errors are found please send suggested corrections or additions through to Science.Officer@mpi.govt.nz.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2017-01A	Research into the demographic parameters for at-risk seabirds as identified by the Risk Assessment (black petrels)	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.	Contracted, in progress	
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.	Approved, not yet contracted	
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.	Approved, not yet contracted	
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.	Contracted	
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.	Approved, not yet contracted	
PRO	PRO2017-08A	Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk	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
		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.	Contracted, in progress	
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. 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.	Contracted, in progress	
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	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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.	Contracted, in progress	
PRO	SEA2017-08	A synthesis of the population work carried out as part of PRO2006-01	Preparation of a report which summerises 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.	Contracted, in progress	
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 iformation on distribution and trends and papers or summaries of biology, age, growth, fecundity and general productivity.	Completed	
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.	Contracted, in progress	
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.	Contracted, in progress	
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 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	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			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.	Contracted, in progress	
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	
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	Report summarising (by fishing method) the bycatch mitigation measures and hurdles to uptake in New Zealand, including development, testing and cost.	Complete	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		Mitigation Measures applicable to New Zealand commercial fisheries			
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 SPRFRMO 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	
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	
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.	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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. 2. To increase the capability of other countries to produce robust estimates of seabird captures.	Contracted, in progress	
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 Hectors dolphins 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 permits 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	
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	
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.	Complete	Richard et al. 2017

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			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.		
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 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.	Complete	Roberts et al. 2016
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	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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. 3. Assess the sensitivity of the risk assessment to live captures.	Complete	Abraham & Richard 2017
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 (translocation) rates. As an alternative, this 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 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.	Complete PRO2014-02.	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			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
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
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.	In progress	Abraham et al. 2016, 2017
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.	Complete	Clement & MacKenzie 2016

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			5. To estimate the distribution of Hector's dolphins along the WCSI in winter 2014 applying an agreed aerial survey methodology.		
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 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	Included in PRO2013-06	
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)
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 and Sagar 2016
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
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

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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; Abraham et al. 2017
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 population viability. 3. As information permits, assess the effect of alternative management strategies.	Complete	Edwards et al. 2017
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, 2013
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
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	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, 2013, 2014; Richard & Abraham 2015

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		in New Zealand commercial fisheries			
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, 2013b, 2013c; 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 lions 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

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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
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
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
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 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.	Complete	MacKenzie & Clement 2014

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			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
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 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.	Complete	Roe 2010b; Roe & Meynier 2012
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

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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; Abraham & Thompson 2009a, 2010, 2011a, 2011b; Thompson & Abraham 2009a; Abraham et al. 2010b
PRO	PRO2007-02	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 et al. 2010a; Thompson & Abraham 2009a, 2009b, 2009c, 2010, 2011; Thompson et al. 2010a, 2010b
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
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

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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, 2010a, 2010b, 2010c; Baker et al. 2008, 2009, 2010
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; Francis 2012; Francis et al. 2015
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, 2011; Smith & Baird 2009, 2011
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.	Complete	Abraham et al. 2010a; Thompson & Abraham 2009a,

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			Recommend mitigation measures appropriate for uptake in non-commercial fisheries in which seabird or marine mammal captures occur.		2009b, 2009c, 2010, 2011; Thompson et al. 2010a, 2010b, 2010c
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 2007a; Baird & Gilbert 2010
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 metric 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 2007b; Baird & Smith 2007b
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 to 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
PRO	ENV2005-09	Data collection to estimate key performance indicators in the	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	Complete	No reports specified as required output

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		Chatham albatross, *Diomedea eremita*.	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.		
PRO	ENV2005-13	Assessment of risk to yellow-eyed penguin Megady-ptes 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 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.	Complete	Fletcher et al. 2008
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

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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 2001102 fishing year, including confidence limits and an investigation of any statistical bias in the estimate.	Complete	Baird 2005a, 2005b; Baird & Doonan 2005
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

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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 sea lion 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
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	To report on incidental captures of non-fish species in New Zealand fisheries.	Complete	Baird 1997

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		fishing operations in New Zealand waters			
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 souther 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
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 estimates 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

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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.		
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.	Contracted, in progress	
NPB	SEA2016-19	Spatial methods for development to support risk assessment (part II). Estimation of capture and retention efficiency for non-target fish species in commercial trawl fisheries	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-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.	Contracted, in progress	
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	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	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		qualitative risk assessment			
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	
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 potential 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
NPB	ENV2013-01	Development of model-based estimates of fish bycatch	1. To develop a statistical modelling approach to estimating 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
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 2012, 2013a, 2013b, 2014a, 2014b; Ballara 2015; Ballara & O'Driscoll 2015.
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 species	Complete	Anderson 2011

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			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, 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
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
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
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
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
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
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, 2011b, 2011c
NPB	TUN2006-02	Estimation of non-target fish catches in	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.	Complete	Griggs et al. 2008

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		the tuna longline fishery	2. To describe bycatch trends in tuna longline fisheries using data from this project and the results of previous similar projects.		
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 20011/2002 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	1. To estimate the catch rates of non-target fish in the 10ngline 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 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.	Complete	Griggs et al. 2007
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	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	Complete	Anderson 2004

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		selected New Zealand fisheries	mackerel (*Trachurus declivis, T. novaezelandiae, & T. symmetricus murphyi*) and scampi (*Metanephrops challengeri*).		
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 fishery, including some general thoughts on the experimental design of field trials.	Complete	Hartill et al. 2006
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 depeadent species, including invertebrates, from deepwater and middle depth fisheries on the Chatham Rise.	Complete	Livingston et al. 2003

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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
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.	Approved, not yet contracted	
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 Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Ongoing	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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.	Ongoing	
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.	Contracted, in progress	
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)
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,	Ongoing analysis	Black et al. 2013; Black & Tilney 2015.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			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.		
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
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
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
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	Complete	Baird et al. 2009, 2011; Baird & Wood 2010;

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			habitat type. 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.		Leathwick et al. 2010, 2012
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, 2010b, 2010c, 2011
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 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).	Complete	Tuck et al. 2010
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.	Complete	Michael et al. 2006

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			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.		
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. 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.	Complete	Tuck et al. 2006a, 2006b
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

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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 effects, and to put the summary in a New Zealand context.	Complete	Cryer et al. 2004
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	1. To determine the effects of fishing on the benthic community structure between North Cape and 1. Cape Reinga.	Complete	Cryer et al. 2000

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		community structure between North Cape and Cape Reinga			
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 to advise the Commission on what management actions may be required to improve the achievement of the objectives for this MPA.	Approved	
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; Williams et al. 2014; Opresko et al. 2014; Clark et al. 2015
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
ECO	SEA2012-17	NPOA Sharks extension work	NPOA Sharks extension work.	Complete	Clarke et al. 2013
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
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 of invertebrates in research trawls and commercial fishing.	Complete	Mills et al. 2013

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ECO	DEE2010-04	Development of a methodology for Environmental Risk Assessments for deepwater fisheries	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
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
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
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; Jones et al. 2016
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 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.	Complete	Hewitt et al. 2014
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
ECO	ENV2009-07	Habitats of particular significance for fisheries	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.	Complete	Morrison et al. 2014d

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		management: kaipara harbour	3. Design and implement cost-effective habitat mapping and monitoring surveys of habitats of particular significance for fisheries management in the Kaipara Harbour.		
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)
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
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
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
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
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
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
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
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
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). 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.	Complete	No reports specified as required output
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.	Complete	Bradford-Grieve & Livingston 2011

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			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.		
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
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	2. To investigate the prediction of year class strength from environmental variables.	Complete	Francis et al. 2005
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.	Complete	Zemke-White et al. 2005

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			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.		
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 lobsterfishing on the environment.	Complete	Breen 2005
ECO	ENV2000-04	Identification of areas of habitat of particular significance for fisheries management within the New Zealand EEZ	1. To review literature and existing data for all significant fish species, including all QMS species, encountered from the 200 1500 m contour 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. 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.	Complete	O'Driscoll et al. 2003
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

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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 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.	Withdrawn	
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	Ongoing	
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.	Contracted	
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 in prep
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
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.	Ongoing	
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.	Ongoing	

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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. in prep
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.	Ongoing	
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.	Ongoing	
ZBD	ZBD2014-09	Climate change risks and opportunities	Objective 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. Objective 2. To prepare a synthesis of the Technical Report that will be informative and provide guidance on what can be done, for stakeholders, policymakers and resource managers.	Ongoing	
ZBD	ZBD2014-10	BPA Biodiversity	Obj 1. To update the inventory of benthic samples and biodiversity data available within BPA and Seamount Closure areas. Obj 2. To process and identify undescribed samples and material in selected BPAs and for selected taxonomic groups. Obj 3. To identify gaps in sample coverage, evaluate priority areas and design a sampling programme to collect appropriate data. Obj 4 To undertake an objective spatial management planning exercise to assess the effectiveness of the current BPAs to protect biodiversity.	Ongoing	
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.	Ongoing	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).	Ongoing	
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	

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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
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	Tier 1 Stat. 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
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	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; McGregor & Horn 2015
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	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.	Complete	Rowden et al. 2013b

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		South Pacific Ocean region.	2. To collate datasets and evaluate modelling approaches which are likely to be useful to predict the distribtuion of vulnerable marine ecosystmes in the South pacific Ocean region.		
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. 2011b
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
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
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 vulnerability of New Zealand species of rhodoliths to environmental stressors.	Complete	Nelson et al. 2012
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	Complete	Gardner et al. 2010

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			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.		
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 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.	Complete	Jones et al. 2016, submitted
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
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	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.	Complete	Law et al. 2012; Boyd & Law 2011

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		climate change induced ocean acidification	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 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
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 Isds 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
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
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 seamounts to an oceanographically distinct location.	Complete	Clark et al. 2009
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	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.

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			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.		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 et al. 2007a, 2009b, 2010, 2013; Schiaparelli

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					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- 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 sea bed 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 (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-	Complete	Bowden 2011; Bowden et al. 2011b, 2014; Bowden & Hewitt 2012; Coleman and Lörz 2010; Compton et al. 2012; Floerl et al. 2012; Hewitt et al. 2011a, 2011b; Lörz 2011a, 2011b; Nodder et al. 2012

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			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.		
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 off shore 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

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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 opportunistically as time allows. 2. To describe the distribution of broad macro epifauna groups (I.D. 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 level. 6. To process and archive biological samples and data into databases and collections for future analysis in meeting the Overall Objectives above.	Complete	Nodder 2008; Nodder et al. 2011
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 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)	Terminated	Smith 2006

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			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 seaRoss 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 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).	Complete	MacDiarmid & Stewart 2015; Mitchell & MacDiarmid 2006
ZBD	ZBD2005-05	Long-term effects of climate variation and human impacts on the structure and functioning of New	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.	Complete	Carroll et al. 2015; Lalas et al. 2014; Lalas & MacDiarmid 2014; Lorrey et al. 2013;

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		Zealand shelf ecosystems	3. To collect additional oral histories from Maori and non-Maori 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 Maori 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.		MacDiarmid et al. 2016a, 2016b, submitted a, submitted b; Maxwell & MacDiarmid 2016; McKenzie & MacDiarmid submitted; Neil et al. 2012; Parsons et al. 2011; Paul 2012, 2014; Pinkerton et al. 2015b; Smith 2011
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; 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. 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.	Complete	Connell et al. 2010; Dunn 2009; Dunn et al. 2009b, 2010a, 2010b, 2010c; Forman & Dunn 2010; Horn et al. 2010; Stevens & Dunn 2010

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			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.		
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
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 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 macrofau na 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.	Complete	Lohrer et al. 2010
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	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, 2013a;

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		communities of the north western Ross Sea			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 section of the Macquarie Ridge Complex that lies between New Zealand and Macquarie Island.	Complete	Grayling 2004
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 2. To provide a molecular description and characterisation of gobies that are introduced (*Arenigobius bifrenatus* and *Acentrogobius pflaumii*) cryptogenic (*Parioglossus marginalis*) or native (eg.*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	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 (eg.*Favonigobius*	Complete	Lavery et al. 2006

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		marine species – gobies.	*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 Sound. 3. To analyse Ross Island Sea-Level data.	Complete	Norkko et al. 2002
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-	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	Complete	Rowden et al. 2012b

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		sediment communities	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.		
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 distniution of macro-invertebrate assemblages on the seamounts. 3. To compare the macro-invertebrate species diversity of neighbouring seamounts. 4. To evaluate and collect samples fiom 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
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

Theme	Project code	Project title	Specific objectives	Status	Citation/s
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 Zealand's Deepwater Benthic Biodiversity	1. To review and document existing published and unpublished reports and data describing New Zealand's deepwater benthic biodiversity. 2. To make recommendations on representative communities and potentially impacted communities that could be the subject of directed research.	Complete	Key 2002
ZBD	ZBD2000-09	Antarctic fish taxonomy	1. Ross Sea fishes processing and identification.	Complete	Roberts & Stewart 2001

19.10 APPENDICES FROM CHAPTER 9 FISH AND INVERTEBRATE BYCATCH

Appendix 19.10.1: 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.
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.

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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.
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*

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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*

Appendix 19.10.1 [Continued]:

								Fishery	
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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*

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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*

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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*
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*

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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
TSQ						0.03		0.13	*Todarodes filippovae*
TTA			0.03						*Typhlonarke tarakea*

Appendix 19.10.1 [Continued]:

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
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 19.10.2: BYCATCH: Total annual bycatch 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
Total									Fishery
Fyr	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
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

Appendix 19.10.2 [Continued]:

									AUCKLAND ISLANDS
Total	Fishery								
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
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	56	0	54	0	0	1 480
2015	329	713	10	56	0	13	50	0	1 171
2016	519	989	10	56	0	13	50	0	1 637

									CHATHAM RISE
Total	Fishery								
Fyr	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	9 879	2 378	141	604	0	16 853
2015	2 261	778	1 041	9 879	2 378	340	553	0	17 230
2016	1 300	2 133	1 041	9 879	2 378	340	553	0	17 624

									COOK STRAIT
Total	Fishery								
Fyr	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
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

Appendix 19.10.2 [Continued]:

									COOK STRAIT
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
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	645	0	0	0	0	656
2015	0	0	11	645	0	0	23	0	679
2016	0	2	11	645	0	0	23	0	681

									EAST COAST NORTH ISLAND
Total									**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	507	0	0	0	0	1 057
2015	332	0	229	507	0	0	54	0	1 122
2016	234	0	229	507	0	0	54	0	1 024

Appendix 19.10.2 [Continued]:

									NORTHLAND
Total									Fishery
Fyr	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	383	0	0	26	0	1 019
2015	479	0	44	383	0	0	16	0	922
2016	318	8	44	383	0	0	16	0	769

									PUYSEGUR
Total									Fishery
Fyr	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
2013	0	66	21	513	0	11	0	0	611
2014	0	127	21	513	0	75	42	0	778

Appendix 19.10.2 [Continued]:

									PUYSEGUR
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2015	0	90	21	513	0	3	68	0	695
2016	0	107	21	513	0	3	68	0	712

									STEWART-SNARES SHELF
Total									Fishery
Fyr	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	2 269	1 135	31	0	0	12 431
2015	0	7 126	514	2 269	1 135	1	0	0	11 044
2016	0	5 625	514	2 269	1 135	1	0	0	9 544

									SUBANTARCTIC
Total									Fishery
Fyr	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
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

Appendix 19.10.2 [Continued]:

									SUBANTARCTIC
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
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	133	0	0	0	40	334
2015	35	7	139	133	0	0	0	40	354
2016	5	4	139	133	0	0	0	40	321

									WEST COAST NORTH ISLAND
Total									Fishery
Fyr	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	51	5 281	0	64	0	5 410
2015	0	0	14	51	5 281	0	30	0	5 376
2016	0	0	14	51	5 281	0	30	0	5 376

									WEST COAST SOUTH ISLAND
Total									Fishery
Fyr	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
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

Appendix 19.10.2 [Continued]:

								WEST COAST SOUTH ISLAND
Total								**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
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	3 827	1 003	0	11	0	5 300
2015	34	450	428	3 827	1 003	0	148	0	5 890
2016	39	0	428	3 827	1 003	0	148	0	5 445

Appendix 19.10.3: 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
Total									**Fishery**
Fyr	**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	828
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	38	0	7	7	0	2 046
2004	270	1 741	12	20	0	3	5	0	2 052
2005	508	1 234	1	14	0	0	1	0	1 758
2006	414	2 574	0	3	0	0	0	0	2 991
2007	448	812	0	3	0	0	0	0	1 263
2008	316	538	0	3	0	0	1	0	858
2009	412	1 331	0	0	0	1	0	0	1 744
2010	520	1 079	12	2	0	0	0	0	1 614
2011	545	1 386	66	0	0	0	1	0	1 998
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	4	0	1	2	0	735
2015	250	389	6	4	0	0	2	0	651
2016	421	1 130	6	4	0	0	2	0	1 563

Appendix 19.10.3 [Continued]:

								CHATHAM RISE
Total								**Fishery**
Fyr	**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	12 824
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	8 566	80	68	478	0	18 598
2004	218	902	806	3 181	5	99	538	0	5 750
2005	2 117	2 156	1 080	870	5	95	520	0	6 843
2006	898	2 681	955	963	271	93	1 399	0	7 260
2007	2 249	2 392	546	1 814	52	46	491	0	7 590
2008	1 417	864	1 016	1 562	47	28	129	0	5 064
2009	1 041	327	875	2 610	11	26	195	0	5 085
2010	1 418	419	950	6 548	3	27	76	0	9 442
2011	1 002	1 083	879	2 685	3	25	29	0	5 706
2012	1 138	395	517	1 293	4	61	115	0	3 522
2013	1 743	105	517	2 628	32	9	71	0	5 105
2014	2 245	29	517	2 628	3	30	18	0	5 470
2015	1 718	478	517	2 628	3	7	32	0	5 384
2016	1 262	1 859	517	2 628	3	7	32	0	6 309

								COOK STRAIT
Total								**Fishery**
Fyr	**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
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	2 570	0	3	3	0	2 585
2004	0	0	15	977	0	0	2	0	994
2005	28	0	31	593	0	0	3	0	656
2006	1	0	36	206	0	2	7	0	252
2007	1	0	48	851	0	0	3	0	903
2008	2	0	35	576	0	0	3	0	617
2009	0	0	8	510	0	0	4	0	522
2010	0	0	7	367	0	0	2	0	376
2011	0	0	8	747	0	0	3	0	758
2012	0	0	5	340	0	0	2	0	347
2013	0	0	5	178	0	0	1	0	184
2014	0	0	5	178	0	0	2	0	185

Appendix 19.10.3 [Continued]:

									COOK STRAIT
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2015	0	0	5	178	0	0	4	0	187
2016	0	2	5	178	0	0	4	0	189

									EAST COAST NORTH ISLAND
Total									Fishery
Fyr	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	223	0	23	0	0	979
2004	161	0	58	116	0	4	5	0	344
2005	585	0	64	41	0	4	7	0	700
2006	247	0	96	51	0	3	56	0	452
2007	470	0	66	60	0	1	59	0	655
2008	289	0	84	38	0	0	59	0	470
2009	186	0	78	48	0	0	83	0	396
2010	601	0	81	89	0	0	100	0	870
2011	423	0	73	147	0	0	22	0	665
2012	209	0	79	56	0	0	32	0	376
2013	238	0	79	104	0	0	57	0	478
2014	301	0	79	104	0	0	37	0	521
2015	285	0	79	104	0	0	13	0	481
2016	255	0	79	104	0	0	13	0	451

									NORTHLAND
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	615	0	23	13	0	0	5	0	656
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	81	0	0	4	0	496
2004	235	2	5	49	0	0	1	0	292
2005	746	0	12	15	0	0	1	0	774
2006	281	4	13	60	0	0	0	0	358
2007	507	5	12	43	0	0	1	0	568
2008	355	1	24	32	0	0	1	0	412

Appendix 19.10.3 [Continued]:

								NORTHLAND
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2009	231	2	11	34	0	0	4	0	282
2010	505	0	13	141	0	0	0	0	659
2011	335	0	19	93	0	0	1	0	448
2012	268	0	23	61	0	0	6	0	358
2013	505	3	23	134	0	0	9	0	674
2014	454	0	23	134	0	0	7	0	618
2015	342	0	23	134	0	0	3	0	503
2016	321	6	23	134	0	0	3	0	488

									PUYSEGUR
Total									Fishery
Fyr	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	196	0	172	18	0	3 667
2004	4	122	34	38	0	15	9	0	222
2005	5	196	39	295	0	15	4	0	554
2006	0	147	54	188	0	11	1	0	401
2007	0	12	45	129	0	3	0	0	189
2008	0	10	116	113	0	11	1	0	252
2009	0	4	14	88	0	2	2	0	110
2010	0	19	4	66	0	1	2	0	92
2011	0	49	57	151	0	3	3	0	262
2012	0	12	16	132	0	0	2	0	163
2013	0	8	16	216	0	0	0	0	241
2014	0	37	16	216	0	2	5	0	276
2015	0	34	16	216	0	0	0	0	267
2016	0	71	16	216	0	0	0	0	304

									STEWART-SNARES SHELF
Total									Fishery
Fyr	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

Appendix 19.10.3 [Continued]:

									STEWART-SNARES SHELF
Total									**Fishery**
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
2003	44	3 383	34	1 035	25	141	0	0	4 661
2004	9	2 858	58	499	4	24	0	0	3 452
2005	3	3 277	70	388	7	23	0	0	3 768
2006	0	4 713	18	1 299	10	37	1	0	6 077
2007	0	1 892	218	434	3	5	0	0	2 553
2008	0	1 096	74	82	1	5	0	0	1 258
2009	0	1 816	34	136	7	2	0	0	1 995
2010	0	1 656	23	277	2	4	0	0	1 962
2011	0	2 690	23	807	2	4	0	0	3 526
2012	0	1 513	252	589	3	4	0	0	2 362
2013	0	1 544	252	615	32	6	0	0	2 449
2014	0	1 152	252	615	17	6	0	0	2 042
2015	0	961	252	615	17	0	0	0	1 846
2016	0	1 284	252	615	17	0	0	0	2 168

									SUBANTARCTIC
Total									**Fishery**
Fyr	**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	1 194
2003	19	497	267	524	0	91	14	250	1 661
2004	13	129	329	209	0	58	19	90	847
2005	0	43	136	107	0	78	5	160	530
2006	0	42	66	21	0	121	27	240	518
2007	0	96	53	47	0	51	4	120	371
2008	0	2	62	122	0	32	4	120	343
2009	0	1	151	11	0	33	6	120	322
2010	0	5	115	20	0	36	3	120	299
2011	0	6	20	15	0	12	2	120	175
2012	0	2	86	1	0	9	0	120	218
2013	0	6	86	12	0	0	0	120	224
2014	14	1	86	12	0	0	0	120	233
2015	25	0	86	12	0	0	0	120	243
2016	4	1	86	12	0	0	0	120	223

Appendix 19.10.3 [Continued]:

									WEST COAST NORTH ISLAND
Total									Fishery
Fyr	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	496
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	47	183	0	6	0	376
2004	0	2	0	9	133	0	7	0	151
2005	5	0	1	2	98	0	15	0	121
2006	0	0	2	13	77	0	11	0	103
2007	0	1	19	6	165	0	2	0	193
2008	0	0	15	14	166	0	13	0	207
2009	0	0	17	14	97	0	3	0	131
2010	0	0	11	8	264	0	6	0	290
2011	0	0	23	9	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	18	165	0	60	0	251
2015	0	0	9	18	165	0	10	0	202
2016	0	0	9	18	165	0	10	0	202

									WEST COAST SOUTH ISLAND
Total									Fishery
Fyr	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
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	2 762	7	0	1	0	3 158
2004	0	9	107	1 655	1	0	1	0	1 773
2005	6	29	165	465	3	0	0	0	668
2006	0	5	153	1 414	29	0	7	0	1 608
2007	0	14	157	524	84	0	1	0	780
2008	0	0	236	626	7	0	0	0	869
2009	0	1	204	479	5	0	0	0	689
2010	0	1	234	480	4	0	0	0	719
2011	0	0	183	1 045	1	0	2	0	1 231
2012	9	0	209	1 029	5	0	3	0	1 256
2013	14	0	209	2 082	7	0	6	0	2 319
2014	19	0	209	2 082	9	0	10	0	2 330

Appendix 19.10.3 [Continued]:

									WEST COAST SOUTH ISLAND
Total									Fishery
Fyr	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2015	21	20	209	2 082	9	0	48	0	2 390
2016	28	0	209	2 082	9	0	48	0	2 377

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