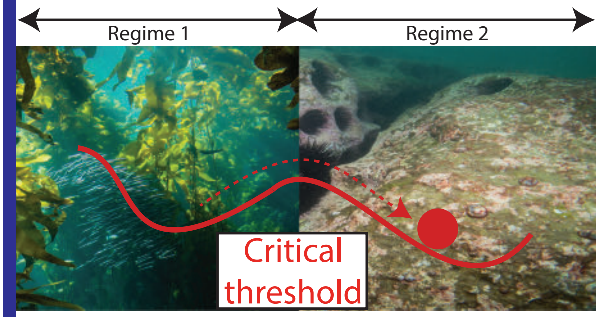
Chapter 13: Trophic and ecosystem-level effects - Technical Summary

Altered food webs, declining fish productivity



Baseline and regime shifts

1. THE ISSUE IN BRIEF

- Marine ecosystems are dynamic, and fluctuate naturally between different states
- Among other causes (e.g. climate change), fishing can create an imbalance by selectively removing species, which are prey and predators of other species in the food-web
- Ecosystems tend to correct any imbalance, but sometimes alterations to the food web functioning are long-lasting
- Understanding, evaluating, or predicting these changes (as well as separating natural and anthropogenic causes) is challenging, but progress has been made

2. DIFFERENT LEVELS OF EFFECTS

- Trophic and ecosystem-level effects can be ranked based on their magnitude and spatial scale
- At the local scale, effects of activities such as offal discard by fishing vessels can promote scavenging by seabirds, which can have a direct effect on the feeding behaviour of some species
- First order effects take place when the predator/prey balance is disrupted
- Second order effects influence many trophic levels, at larger scales, and are also known as trophic cascades
- Large-scale effects are called regime shifts, and can affect whole ecosystems, occasionally leading to a permanently altered ecological state

3. EXAMPLES OF FISHERIES-INDUCED CHANGES

Fishing pressure may alter fish populations

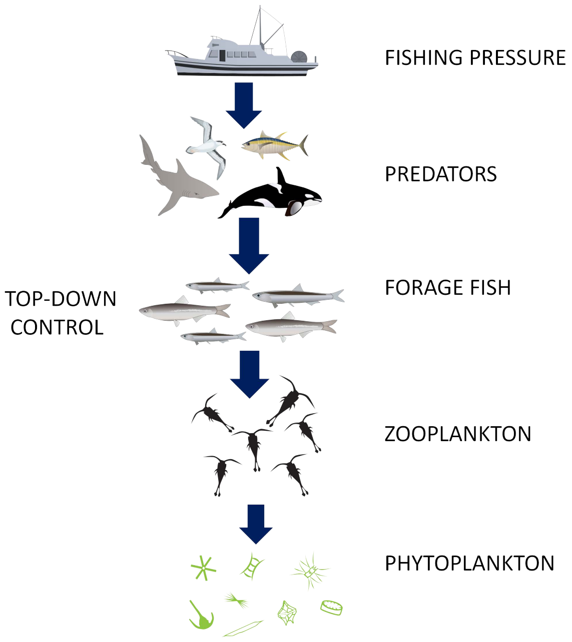
- demographic and genetic structure
- spatial distribution
- life-history traits (e.g. age at maturity)

as well as habitat structure and trophic structure of the food web, depending on fishing pressure and extent

Fishing typically affects food webs through:

- Top-down control (e.g. predator removal)
- Middle-out control (e.g. small demersal fish removal, thus altering nutrient transfer)

- Selective fishing pressures on key components of the food web are more likely to have significant effects
- Detection and scale of effects is a complex issue, and
- The use of multiple indicators can separate them from natural fluctuations



Schematic of fisheries-induced top-down effects on the ecosystem

13 TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

Status of chapter	This chapter has been updated for AEBAR 2024.
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), habitat and other 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	• Developing an ecosystem approach to fisheries management that incorporates trophic and ecosystem-level effects into fisheries management decisions. • Understanding how fishing interacts with other stressors in marine ecosystems.
MPI research (latest)	Publications: • AEBR 324 "Fishery-induced trophic cascades and sea urchin barrens in New Zealand: A review and discussion for management" • AEBR 301 "Role of low and mid-trophic level fish in the Hauraki Gulf ecosystem" Research: • ZBD2023-03 "Summarising and updating knowledge on the distribution of kina barrens in key regions of New Zealand" • INV2023-10 "Ecosystem based management: a case study on the east coast of the South Island" • SEA2023-08 "Implications of incorporating ecosystem considerations into management targets and limits for fish stocks" • ZBD2023-05 "Importance of macroalgae to New Zealand fisheries and the effects of climate change"
NZ government research (current)	• NIWA SSIF (Strategic Science Investment Fund) – Coasts & Oceans centre: Multi-use Marine Ecosystems • Government, NIWA SSIF- Marine food webs: Marine food webs | NIWA
Related chapters	• Benthic (seabed) impacts (Chapter 11) • Climate and oceanographic context of New Zealand fisheries (Chapter 12). • Land-based effects on fisheries (Chapter 15). • Marine biodiversity (Chapter 18). • HPSFM (Chapter 14)

13.1 OVERVIEW

This chapter describes the trophic and ecosystem-level effects which may arise from fishing or in combination with other drivers of change on marine ecosystems in the New Zealand region. 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.

Identifying the trophic and ecosystem-level effects of fishing and incorporating them into fisheries management decisions is essential to advance New Zealand's ecosystem approach to fisheries management. Summaries of the impact of fishing on trophic and ecosystem function for individual species can be found in the Environmental and Ecosystem considerations section of selected chapters of the annual Fisheries New Zealand assessment plenary (Fisheries New Zealand, 2024a). This chapter provides an overview of the potential types of trophic and ecosystem-level effects of fishing in the marine environment with descriptions of relevant examples from within New Zealand where available. While these descriptions are currently used in a qualitative manner to inform sustainability round reviews for fisheries species, Fisheries New Zealand is exploring ecosystem modelling approaches to better evaluate and incorporate the impact of different management approaches on the wider ecosystem, and to identify critical data gaps required to use these models (see Project Summary Table in the chapter introduction).

The need to consider these effects in fisheries management decisions is set out in New Zealand legislation, current New Zealand government strategic/operational policies, and international best practice. Relevant legislation, policies and best practices for managing the effects of fishing on the aquatic environment are summarized in Chapter 1 of the AEBAR (Sections 1.2 and 1.3). The requirement to incorporate trophic and ecosystem-level effects in fisheries management decisions is referenced in multiple documents including:

- The Fisheries Act 1996 (*https://www.legislation.govt.nz/act/public/1996/0088/latest/DLM394192.html*)
- MPI's Strategy 'Our Strategy': to grow the sustainable use of our natural resources (*http://www.mpi.govt.nz/about-mpi/our-strategy*).[1]
- FAO Code of Conduct for Responsible Fisheries (*https://www.fao.org/3/v9878e/v9878e00.html*)
- Marine Stewardship Council (MSC 2010) Principle 2
- The Chief Science Advisor's report to the Prime Minister on The Future of Commercial Fishing in Aotearoa (2019) (*https://www.pmcsa.ac.nz/topics/fish/*)
- The Te Mana o Te Taiao (Aotearoa New Zealand Biodiversity Strategy 2020) (*https://www.doc.govt.nz/globalassets/documents/conservation/biodiversity/anzbs-2020.pdf*)
- The 2022 New Zealand High Court decision on the management of spiny lobster in CRA1 (*https://www.mpi.govt.nz/dmsdocument/55012-2022-High-Court-judgment-decision-for-Northland-rock-lobster*)
- The International Tribunal for the Law of the Sea (Advisory Opinion on Climate Change, 2024)

13.2 TROPHIC AND ECOSYSTEM EFFECTS

13.2.1 DEFINITION OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

'Trophic effects' are changes to the structure and function of food webs occurring entirely or largely because of changes in the feeding of organisms within an ecosystem. 'Ecosystem-level effects' are defined as changes occurring across several trophic levels. An ecosystem is defined as a biological community of interacting organisms and their physical environment. 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 temporal and spatial scales, it is often difficult to be sure of the magnitude of the change or the underlying cause(s).

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[1] Ministry for Primary Industries. Our Strategy. http://www.mpi.govt.nz/about-mpi/our-strategy.

This has led to much speculation and disagreement as to the mechanism or processes involved, and what management should have done to prevent ecosystem-level effects or should do to respond to the changes once they have 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 (Schroeter et al. 1993). Internationally, trophic and ecosystem-level effects is an active area of research, and one in which there are generally more hypotheses than well-accepted empirical demonstrations of the effects. The following sections describe the types of trophic and ecosystem-level effects observed in the marine environment, the contributing factors and scales over which they occur, and references to relevant examples within New Zealand.

13.2.2 TYPES OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

13.2.2.1 FIRST ORDER TROPHIC EFFECTS: PREY AVAILABILITY AND PREDATION RELEASE

Changes to the abundance, size structure and functional type[2] of a species can affect both its predators and prey through trophic interactions (Rosas-Luis et al. 2017). Increasing the abundance of a prey species may positively affect its predators (because they can expend less energy 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 (Lynam et al. 2017, Durante et al. 2022). 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; Eger & Baum 2020). 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.

For example, 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 and Canada's East Coast dropped by a factor of five, from more than 150 000 metric tonnes (MT) to about 30 000 MT (Mayo et al. 1998, Steele et al. 1992). 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 MT (NEFSC 1998). 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, it is hypothesised that a reduction in the abundance of a piscine predator by fishing led to an increase in the prey species – a large-scale 'predation release'.

13.2.2.2 SECOND ORDER TROPHIC EFFECTS: 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, (Schiel 2013, Pershing et al. 2015, Schultz et al. 2016, Spyksma et al. 2017, Kerr & Grace 2017, Eger & Baum 2020, Williams et al. 2021, Wing et al. 2022a). 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). Although initial research on trophic cascades focused on freshwater ecosystems, the potential for

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[2] '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.

fishing to cause trophic cascades in marine ecosystems is thought to be as strong or stronger than in freshwater ecosystems (ICES 2005, Borer et al. 2005). In New Zealand, a trophic cascade is thought to have occurred due to overfishing of the predators of the native sea urchin or kina (*Evechinus chloroticus*), which resulted in overgrazing and loss of kelp on shallow coastal reefs (Doheny et al. 2023, see Section 13.4.2 Case Study 2 for a full description). The evidence for this trophic cascade is based on observations from marine reserves in northeastern New Zealand, however the strength of this trophic effect and whether or not it is occurring in other parts of New Zealand remains uncertain.

A well-recognised international example of a top-down trophic cascade is between sea otters (*Enhydra lutris*), kelp grazing sea urchins, and kelp and seagrass (*Zostera marina*) in the north-east Pacific. The near extinction of sea otters due to hunting in the eighteenth and nineteenth centuries allowed urchin populations to increase, leading to overgrazing of kelp and seagrass beds (Estes & Palmisano 1974, Szpak et al. 2013, Gregr et al. 2020). Protection of sea otters and subsequent expansion or reintroduction of sea otter populations into their former range reversed this cascade (Estes & Palmisano 1974, Estes 1996, Estes & Duggins 1995). The cascading effects of sea otters on kelp extended to other coastal marine species including fish (Markel & Shurin 2015), gulls (Irons et al. 1986), bald eagles (Anthony et al. 2008) and Stellar's sea cows (Estes et al. 2016). However, the generality of the sea otter-urchin-kelp cascade has been questioned. For example, based on experimental treatments, Carter et al. (2007) and Raymond et al. (2021) found that the occurrence of this trophic cascade differs spatially due to environmental and biological variability. Kelp re-colonisation was much faster in southeast Alaska than the Aleutian Islands because sea urchins were smaller (and therefore ate less kelp) and had fewer recruitment events (population growth was slower) over the same timeframe (Estes & Duggins 1995). These observations suggest that trophic cascades are not always generalisable across spatial scales and local environmental and ecological conditions should be taken into account.

13.2.3 PERSISTANCE OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

13.2.3.1 REGIME SHIFT

An ecosystem can change from one state to another when disturbances push it past a tipping point – this transition is called a regime-shift (Aebischer et al. 1990, Estes & Duggins 1995, Beaugrand et al. 2002, Daskalov et al. 2007, Wernberg et al. 2016, Vollset et al. 2022). Regime shifts can occur over large scales, affect many parts of the ecosystem and may be hard or slow to reverse ('hysteresis') (Figure 13.1). Ecosystem-level restructuring may also maintain the system in its new state by means of negative feedback loops (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 fish stocks. For example, the North Sea has historically been a hotspot for both overfishing and climate impacts (rapid warming and acidification) resulting in regime shifts. Between 1958 and 1999, the North Sea experienced two changes in hydro-climatic forcing that caused rapid, temperature-driven ecosystem shifts: a cold-biological shift (1962–1982) and a warm-biological shift (1984–1999) followed by a new dynamic warm regime after the late 1980s that favoured jellyfish in the plankton and echinoderms in the benthos (Kirby et al. 2008, 2009). Some stocks of Atlantic Cod (*Gadus morhua*), a top predator that was formerly widely distributed throughout the North Sea, experienced a collapse over the same timeframe. Atlantic Cod require cooler temperatures for successful spawning (< 10 °C), with early life stages dependent on copepods and adults feeding mainly on decapods/crustaceans (Kjesbu et al. 2023). The collapse of Atlantic cod stocks in the North Sea and central Baltic Sea is attributed to the negative effects of the regime shift on prey availability and cod reproduction (Voss & Quaas 2020) in combination with overfishing (Reid et al. 2003, Beaugrand 2004, Weijerman et al. 2005, Casini et al. 2008, Möllmann et al. 2008, Lindegren et al. 2010).

13.2.3.2 POTENTIAL FOR RECOVERY

Trophic and ecosystem-level effects of fishing may affect the ability of fish stocks to recover (rebuild) following over-exploitation. While species that mature early in life and are fished with size-selective equipment may recover more easily, many species experience little or no recovery up to 15 years following over-exploitation (Hutchings 2000). The failure of many species to recover is unexpected based on their reproductive capacity and habitat availability, and suggests that other factors are at play. Some researchers hypothesise that there are ecosystem-level impacts of over-exploitation, such as changes to community structure or habitats that inhibit fisheries from rebuilding despite management action (Steele & Schumacher 2000).

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 consume 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 real-world example is described in Casini et al. (2008), based on a 33-year time series in the Baltic Sea, that showed that the reduction of the cod population by fishing led to increases in abundance 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 hypothesized to be a significant factor slowing the recovery of the cod population (Jarre-Teichmann et al. 2002, Köster et al. 2003a, 2003b, Casini et al. 2009).

The prevalence of trophic or ecosystem-level effects slowing or stopping recovery after fisheries collapses has been disputed. For example, part of the same cod stock discussed above (eastern Baltic cod) showed rapid recovery after more than 20 years of low biomass and productivity. Based on increases in biomass, Cardinale & Svedäng (2008) concluded that the recovery was driven by a sudden reduction in fishing mortality rather than climate or ecosystem effects, and that single species regulation remained a suitable approach in handling natural resources. However, following the reported recovery, substantial increases in fishing quotas for cod from 2009–2014 paired with declining biological parameters resulted in a collapse of the stock to unprecedented levels (Eero et al. 2023). This case study suggests that environmental ecological conditions in the 1970's–1990's benefitted the Baltic Cod stock more than could be explained by recruitment alone and emphasizes that careful consideration of long time series that combine several indicators will provide a more comprehensive understanding of stock status and allow for more realistic target setting.

There is growing consensus among fisheries managers globally that recovery and effective management of fisheries requires a combination of traditional measures (quota allocation, catch settings) alongside strategic fishing closures, selective fishing gear and economic incentives (Worm et al. 2009). Linking these approaches depends on collaboration between fisheries managers, scientists and conservation biologists to use the best available data to manage for sustainable harvesting within ecosystem and environmental limits. In New Zealand, the Crown also must provide protection for and scope for exercise of rangatiratanga (sovereignty) in respect of traditional fisheries, as customary fishing rights are guaranteed to tangata whenua under Te Tiriti o Waitangi and protected by the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992 and the 1992 Deed of Settlement. This requires the development of equitable partnerships that include collaborative and participatory management of tangata whenua (Department of Conservation 2022, Joseph et al. 2019, Hera Maxwell 2019).

13.3 CAUSES OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

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 ocean 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. Key sectoral human impacts on marine ecosystems in New Zealand include fishing and land-use impacts associated with coastal development, farming and forestry. The cumulative effects of cross-sector impacts and climate change to coastal ecosystems are recognised in parts of New Zealand such as the Marlborough Sounds (Urlich & Handley 2020). This chapter will mainly focus on the trophic effects of the fishing sector, but further information on the impacts of land-use on our fisheries is discussed further in AEBAR Chapter 15 Land-based effects on the coastal environment. Further discussion of 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 (Behrens & Lafferty 2004, Harvell & Lamb 2020), from the arrival of non-indigenous invasive species (Mead et al. 2013, Kotta et al. 2018) 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.

13.3.1 ENVIRONMENTAL AND CROSS SECTOR-DRIVEN CHANGE

Marine ecosystems are impacted by environmental factors such as climate (Fasham et al. 2001, Schiermeier 2004, Frank et al. 2007, Mackinson et al. 2009), sedimentation (Morrison et al. 2009), invasive species (ICES 2005, D'Amen & Azzurro, 2020) and pollution. Climate forcing occurs over various timeframes, from seasonal periods, to 1–3 year oscillating but erratic periods (including marine heatwaves), 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). Internationally and domestically, there is increasing concern about the current and potential future impacts of climate change on fisheries (IPCC 2007a, 2007b, Valdes et al. 2009, Rice & Garcia 2011, Barange et al. 2018, Cheung & Frölicher 2020). In addition to long-term warming, climate change impacts include marine heatwaves (Cheung & Frölicher 2020), increased ocean acidification (Law et al. 2018), increased climate variability (Collins 2000), deoxygenation (Kim et al. 2023) which can affect primary production (Pinkerton et al., 2024a). Sedimentation is an important land-based environmental effect on New Zealand's fisheries, impacting both the species themselves and smothering important nursery and feeding habitats (Morrison et al. 2009). Changes to nutrient loading to the coastal zone from land run-off are also important for water quality and coastal ecosystems (Fraser et al., 2021). Excess nutrient loads can lead to increased algal blooms, reduced oxygen levels, and potentially eutrophication, which can impact marine ecosystems (Plew et al. 2020, Snelder et al. 2020). Environmental change can also make it easier for invasive species to affect New Zealand ecosystems (Hayden et al. 2009, Champion, 2018).

There are many examples internationally of trophic and ecosystem-level effects occurring from environmental change affecting marine food webs (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004). In the eastern Pacific abrupt declines in local primary and secondary production can be caused by El Niño/Southern Oscillation (ENSO) events (Barber & Chavez 1983, Pearcy et al. 1985, Arcos et al. 2001, Hollowed et al. 2001, Arnés-Urgellés et al. 2021). During these ENSO events, the production of small pelagic fishes can be drastically reduced due to the reduction in upwelling of nutrient and plankton-rich water, on which pelagic fishes feed. (Barber & Chavez 1983, Rothschild 1994, McIlvaine 2020), As a result, 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 (Farchadi et al. 2019, Lehodey et al. 2020, Arnés-Urgellés et al. 2021). Historically ENSO events were periodic and ecosystem consequences were short or medium-term. However, climate change appears to contribute to more frequent, extreme and long-lasting El Nino events (Gan et al. 2023), with potentially adverse effects on ecosystem resilience.

Marine heatwaves can also effect marine food webs. For example, marine heatwave events in California between 2014 and 2016 caused sea surface temperatures (SST) to increase by approximately 2.5–7° Celsius above average (Gentemann et al. 2017, Rogers-Bennett & Catton, 2019). The extreme and extended rise in water temperature suppressed spore production and growth of bull kelp *(Nereocystis luetkeana)*, decreasing canopy cover by approximately 90% across 350 km of coastline. Red abalone *(Haliotis rufescens)*, which feed predominantly on bull kelp, were found to be starving and suffered a 43–96% decline by 2018 (Rogers-Bennett & Catton, 2019), prompting an emergency closure of the fishery.

Specific examples of environmental-driven change resulting in trophic or ecosystem-level impacts on fisheries in New Zealand are described in Section 13.4.1 (Bottom-up forcing). An overview of climate impacts on New Zealand's fisheries are described further in AEBAR Chapter 12 and the impacts of sedimentation and other land-based effects on fisheries are discussed in AEBAR Chapter 15.

13.3.2 FISHERIES-DRIVEN CHANGE

Fishing can modify marine food webs through extraction, gear use and fisher behaviour, and may result in ecosystem level impacts. 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 (Dayton et al. 1995; Durante et al. 2021; Durante et al. 2022). Fishing can also have 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 (Montevecchi, 2023), (b) the alteration of fish behaviour and genetic composition (Palkovacs et al. 2018; Hočevar & Kuparinen, 2021) and (c) the modification of the benthos by fishing gear (Kaiser, 2019; Hiddink et al. 2019).

Overfishing of upper trophic level species often leads to a release in predation pressure on mesopredator[3] prey species and in turn, affects lower-level species which mesopredators consume (Myers et al. 2007, Eklöf et al. 2020). Marine ecosystems can be resilient to large trophic changes, but there are limits to this resilience. Ecosystem-level changes (regime shifts) have been observed in very highly impacted (highly overfished/collapsed) systems, mainly due to trophic effects (Estes & Duggins 1995, Bakun & Weeks 2006, Daskalov et al. 2007). For example, in the Black Sea, overfishing of pelagic predatory fish, combined with the introduction of an invasive species (the comb jelly, *Mnemiopsis leidyi*) resulted in a large-scale sustained impacts on the ecosystem including reduced productivity (Daskalov et al. 2007).

Food subsidies, such as offal and discards from fishing vessels can become important sources of food for some marine species (Heath et al. 2014), influencing population dynamics, community composition and interactions, affecting and influencing ecosystem structure (Fondo et al. 2015). In New Zealand, fishery discards are thought to support the diet of seabirds such as the southern Buller's Albatross during breeding on The Snares (James & Stahl 2000) and can influence foraging patterns of Westland Petrels (Waugh et al. 2018). Discards and offal from fisheries are 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. 2009, Forman & Dunn 2012). Anderson & Smith (2005) estimated that between 1999 and 2003, 11 000–14 000 t per year of non-commercial species and 600–2100 t per year of hoki were 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 time as evidence of climate variation, namely a shift in the prevalence of 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 changes in hoki recruitment was investigated and was found not to be supported empirically (Francis et al. 2006, Bradford-Grieve & Livingston 2011). Nonetheless the indirect effects of fishing on the wider ecosystem are important to take into account in order to interpret the overall impact of fisheries on ecosystem function.

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[3] A mesopredator is a predator that occupies a middle trophic level in a food web, regulating species too small for larger predators.

Durante et al. (2020) studied changes in the trophic level of fish landed off the east coast of New Zealand's South Island, from 1931 to 2015. This region is important for both commercial and recreational fishing and has been subject to the highest trawling efforts in New Zealand's EEZ since the expansion of commercial fishing (Baird et al. 2011, Macgibbon & Mules 2023). Analysis of fishery data showed that the weighted average trophic level or Marine Trophic Index (MTI) of landed fish increased gradually from 1931 as fishing intensity increased and more species were landed, peaking around the year 2000 then decreasing (Durante et al. 2020, Durante et al. 2021). The decreased MTI since 2000 illustrates smaller catches made up of lower trophic level species. These patterns are consistent with trends in the global fishing expansion since the 1950s, characterised as "fishing down the food web" (Swartz et al. 2010, Durante et al. 2020).

13.3.3 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, 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). These ecosystem level effects can occur even when fish stocks are not characterised as "overfished", hence the global push for ecosystem considerations to be incorporated more explicitly into fisheries management decisions. Although effective single species management is designed to take some of the effects described in this section into account, a lack of data availability on different life history stages, spatial distribution, bycatch, fish biology, ecology and fisher behaviour can inhibit effective management. Therefore, fishing is hypothesised to strengthen the relationship between environmental forcing and ecosystem variability and reduce ecosystem resilience and ecosystem integrity (as defined in Table 13.1) (e.g. Weijerman et al. 2005, Möllmann et al. 2009, Kirby & Beaugrand 2009 and Ottersen et al. 2006). For example, in Tasmania, overfishing of reef-based predators of sea urchins including spiny lobster (*Jasus edwardsii*) has reduced the resilience of kelp forests to the climate-induced expansion of populations of long-spined sea urchins (*Centrostephanus rodgersii*) and increased the risk of reefs shifting to unproductive sea urchin barrens (Ling et al. 2009). In this case, the combined effects of overfishing and climate change has reduced the resilience of these ecosystems to environmental variability and regime shift.

In northern New Zealand, fishery assessments suggest that spiny lobster stocks in the CRA 1 quota management area (QMA) from Kaipara harbour around to Te Arai Point have been managed near to their estimated maximum sustainable yield since entering the quota management system in the 1990s (Fisheries New Zealand 2023). However, in November 2022 the High Court ruled that catch settings for spiny lobster must consider the wider impact of spiny lobster fishing on the aquatic environment, (specifically pertaining to kina barrens and trophic cascades)[4]. This ruling, and subsequent decisions to further reduce the TAC for CRA1 shows that a precautionary approach that goes beyond MSY may be required for incorporating ecosystem considerations into management decision-making. Regional management approaches that take into account environmental variability within the QMA will also be important (e.g. since kina barrens are prevalent on the east coast of CRA1, but are not considered to be an issue along the west coast).

The effects of fishing that may lead to reduced ecosystem resilience and environmental impacts (Figure 13.1) 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) (Hsieh et al. 2010). Fishing often selects the largest oldest fish removing females that produce the largest quantity and highest quality eggs (Calduch-Verdiell et al. 2014). This increases variability in the abundance of fished species (Hsieh et al. 2006, Hočevar & Kuparinen, 2021).

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[4] The Environmental Law Initiative v Minister for Oceans and Fisheries [2022] NZHC 2969 [11 November 2022)

- **Alteration of spatial structure.** The spatial structure of marine fish populations can encompass a wide range of configurations, including patchy populations, networks, and meta-populations (Kritzer & Sale 2004). Removal or reduction 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.** Size-selective fishing can decrease growth rates and age-at-maturity (Law 2000, de Roos et al. 2006, Hočevar & Kuparinen, 2021), accelerating the response of populations to climate forcing (Perry et al. 2010).
- **Alteration of habitat structure.** Changes to benthic habitat by the direct effects of fishing, particularly bottom trawling and dredging, may lead to a reduction in ecosystem resilience (Althaus et al. 2009, Lambert et al. 2014, Pusceddu et al. 2014).
- **Alteration of ecosystem trophic structure.** Fishing can decrease the stock sizes of predators, which can shift ecosystems from top-down to bottom-up control (Durante et al. 2021). Changes to trophic structure by fishing are predicted to increase ecosystem variability and reduce resilience (Jackson et al. 2001, Perry et al. 2010, Dunn et al. 2017). Ecosystems with multiple weak trophic links and generalist predators are less vulnerable to shifts caused by strongly interacting species (Shin & Cury 2001, Polunin & Pinnegar 2002, Rooney et al. 2006, McCann & Rooney 2009, Johnson et al. 2014).

13.3.4 OTHER POTENTIAL IMPACTS

In addition to environmental factors, other impacts to the marine environment that could interact with fishing to affect ecosystems include invasive species, microplastics and bioaccumulation.

13.3.4.1 INVASIVE SPECIES

Species that are novel or introduced to a new ecosystem also have the capacity to alter trophic cascades and consequently trigger a regime shift. Fishing may enhance the vulnerability of marine ecosystems to invasion by non-native species. For example, in the Mediterranean Sea the loss of native seagrass meadows by bottom-impacting fishing methods, has been hypothesised to increase colonisation success of invasive alga *Caulerpa racemosa* (Kiparissis et al. 2011). International fishing vessels, may themselves also present vectors for the introduction of non-native species that can be found in fouling on hulls, bilge water or on marine equipment. While there are many invasive marine species in New Zealand (https://marinebiosecurity.org.nz/) there are no specific examples of these species directly impacting fisheries or fisheries enabling their spread. Between 2021 and 2023, two species of exotic Caulerpa seaweed (*Caulerpa brachypus* and *Caulerpa parvifolia*) were found in several locations around the upper North Island. This seaweed forms dense mats from 2 to 35 metres depth and poses a risk to marine life central to cultural, recreational and commercial activities. Substantial effort and funds have gone into controlling and removing existing Caulerpa to reduce the potentially large ecological and cultural impacts. A monitoring programme to assess the impacts of exotic Caulerpa on the environment and taonga species was established at Aotea/Great Barrier Island in 2023 (Middleton 2023). However, the enduring impact of *Caulerpa* on New Zealand's fisheries and coastal ecosystems remains unknown.

13.3.4.2 PLASTIC POLLUTION AND BIOACCUMULATION

With approximately 8 million tonnes of plastic entering the ocean annually and affecting approximately 690 species of all sizes, from plankton to large cetaceans (Jambeck et al. 2015, Carbery et al. 2018), plastic pollutants in the marine environment can pose a serious risk to sea life. Microplastics, most often defined as particles less than 5 mm (Arthur et al. 2009), are accessible for uptake to a wide range of organisms, particularly nonselective filter feeders. Depending on size, ingested plastic can either sit in the digestive tracts of animals or can further break down into harmful chemicals and be absorbed by the organism (Carbery et al. 2018), potentially affecting population dynamics (Wu et al. 2022). While sea life may be affected both directly by plastic and by bioaccumulation of microplastics through trophic levels (Miller et al. 2020), there is little to no empirical evidence of wide-spread trophic cascade effects or regime shifts due to plastic pollution. The Continuous Plankton Recorder (see description in 13.6.1.3) study analysed micro-and meso-sized plastics and found that the mean number of plastic pieces in near-surface seawater between New Zealand and the Ross Sea increased 11-fold between late 2009 and 2023, to an average of about 29 pieces per m^3. Almost all (>93% on average) of the plastics were microfibres. Pinkerton et al. (2024b) reports that plastic had a statistically significant and negative association with the abundance of many zooplankton groups, especially at concentrations above about 38 pieces per m^3, although cautioned that it is not known if this negative association between plastic and zooplankton is causal or merely correlative.

Table 13.1: Ecosystem resilience.

Fishing and other factors can impact **Ecosystem resilience,** the amount of disturbance that can be absorbed before an ecosystem shifts to an alternate state (Pimm 1982, Holling 1973, Cohen et al. 1990, Walker et al. 2004). It is difficult to quantitatively measure ecosystem resilience, but three qualitative measures of ecosystem resilience are:

- 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 disturbed? Stable (resilient) food webs can absorb more disturbance without undergoing wholesale reorganisation, tend to have low tendency for ecosystem-level trophic cascades (food web perturbations remain local) and have short recovery times (Walker et al. 2004).

Trophic and ecosystem impacts to the marine environment can also impact ecological integrity (as defined in the National Policy Statement for Indigenous Biodiversity (Ministry for the Environment 2023): the extent to which an ecosystem is able to support and maintain its:

a) composition (being its natural diversity of indigenous species, habitats, and communities); and
b) structure (being its biotic and abiotic physical features); and
c) functions (being its ecological and physical processes).

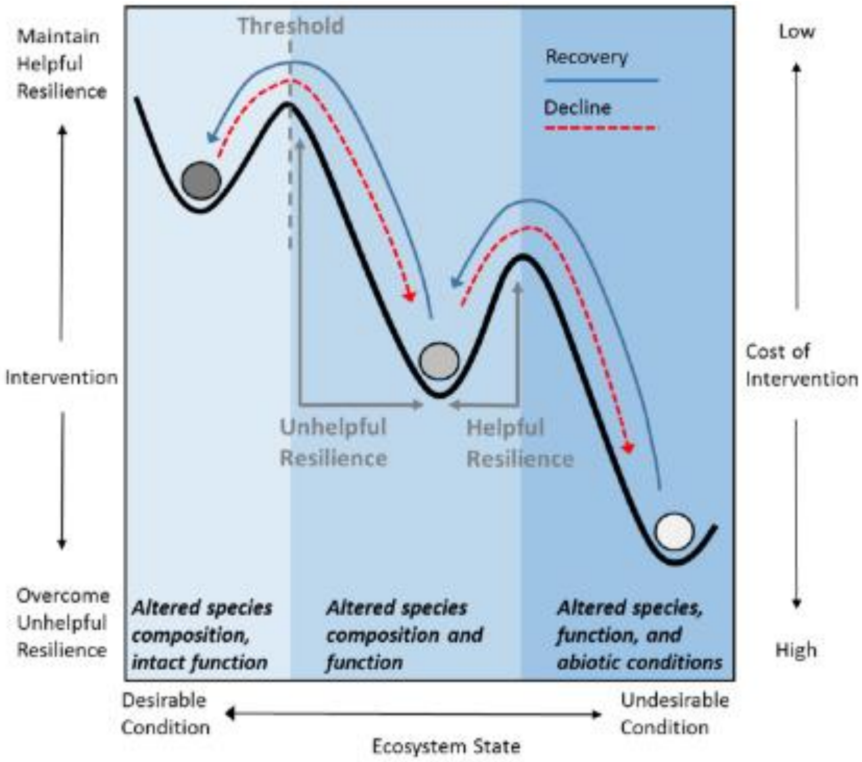

Figure 13.1: A ball and cup figure representing ecological resilience and changes in ecosystem characteristics (image from: Rasmussen & Orr 2017). Once an ecosystem is pushed past a certain threshold (due to changes in species composition and ecosystem function) it can reach a state of hysteresis, or an alternative stable state that cannot be reversed without extensive and costly intervention (Sguotti et al. 2019).

13.4 MECHANISMS DRIVING TROPHIC AND ECOSYSTEM EFFECTS IN NEW ZEALAND

The trophic and ecosystem-level effects of fishing depend on the overall type of ecosystem forcing structure. Most ecosystems will be a mixture of three types: bottom-up, top-down and middle-out forcing. These structures are described in Table 13.2 and illustrated in Figure 13.2. While these processes are often at work in the same ecosystem, and may change over time, identifying dominant patterns of ecosystem behaviour can help to predict or explain the types of change that can result from the combined effects of fisheries harvesting, climate variability/change and other human stressors (Rice 2001).

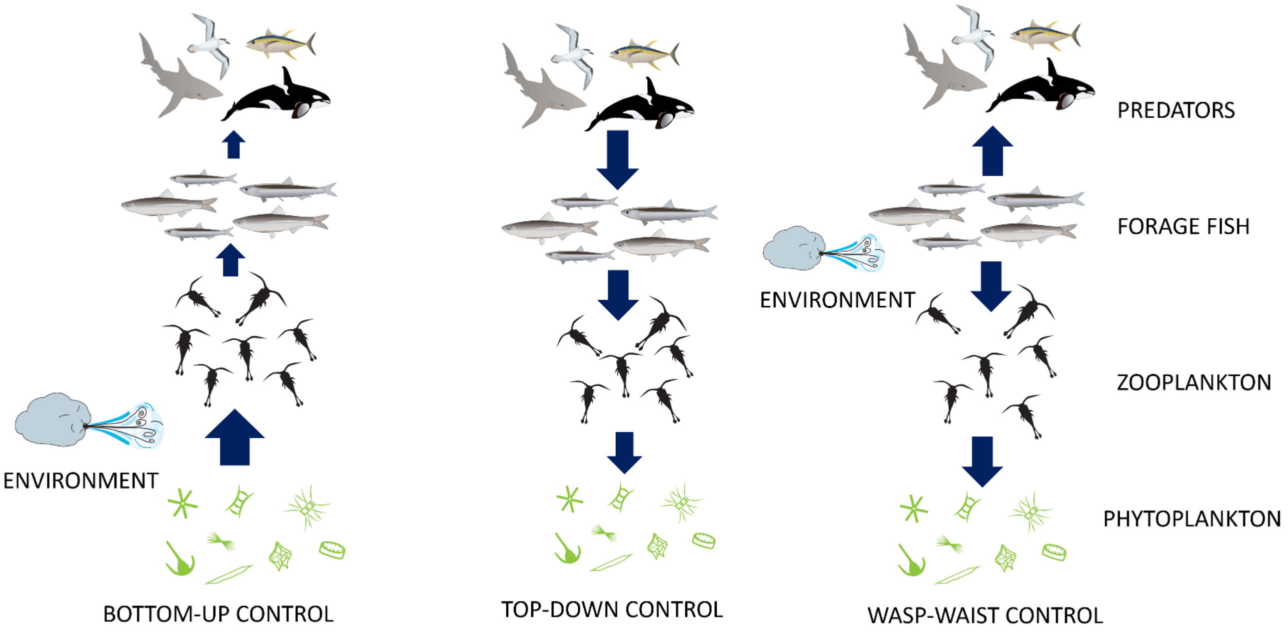

Figure 13.2: Schematic of overall types of ecosystem-forcing (adapted from Gaichas 2019). Symbols courtesy of the Integration and Application Network, ian.umces.edu/symbols/.

Table 13.3: Overall types of ecosystem-forcing.

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. Environmental change is a strong driver of bottom-up forcing in marine ecosystems. On temperate coastal rocky reefs, macroalgae are susceptible to extreme and sustained changes in sea surface temperature (SST) such as marine heatwaves (MHW) and El Niño events (Rogers-Bennett & Catton 2019, Beas-Luna et al. 2020, Tait et al. 2021). Tait et al. (2021) found that giant kelp (*Macrocystis pyrifera*) along the south/southeastern coast of New Zealand declined in abundance and surface canopy area following the extreme marine heatwave (MHW) during 2017/2018. There was also a significant interaction between temperature and turbidity, showing that temperature-induced kelp loss was greater when water clarity was poor. The degradation of kelp communities can affect other parts of the food web, including fish. Bottom-up forcing can also alter the abundance and distribution of organisms and reduce ecosystem resilience to fishing. Poleward shifts in species' distributions are expected to be one of the principal responses to climate change as sea temperatures increase (Cabrelli et al. 2015).
Top-down ecosystem forcing	An ecosystem is said to show top-down forcing if it responds strongly to changes in the abundance of top predators (seabirds, marine mammals, high trophic level fishes, turtles). Fishing is a common top-down force affecting coastal ecosystems. Our 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 (Dunn & Hovel, 2020). 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. This can be observed in marine reserves, which provide refuge for some top predators. For example, the Fiordland marine reserves support higher abundances of two key predators, spiny lobster and blue cod (*Parapercis colias*), with more stable community structure and food webs (Wing & Jack 2014).

Middle-out forced (wasp-waisted) ecosystem	Middle out, also known as “wasp-waist” forced 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 systems dominated by small pelagic fishes that transfer energy from the plankton to larger predators (Rice 1995, Cury et al. 2000, 2004, Bakun 2004, 2006). Ecosystems with wasp-waist control are typically highly productive coastal or upwelling systems with relatively short food chains. However, wasp-waist control can occur in open ocean ecosystems including with species such as herring (*Clupea harengus*), sprat (*Sprattus sprattus*), and anchovy (*Engraulidae spp*) in the North Atlantic (Gifford et al. 2010, Fauchald et al. 2011), krill in the Antarctic (Ainley & Pauly 2014, Atkinson et al. 2014) and copepods (*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.

13.4.1 CASE STUDY 1: BOTTOM-UP ENVIRONMENTAL FORCING ON COASTAL REEFS

Bottom-up Environmental impacts on blue cod

Figure 13.3 Blue cod (*Parapercis colias*)

Adult blue cod (*Parapercis colias,* Figure 13.3) are generalist predators that mainly forage on kelp-associated fish and crabs in coastal waters around much of New Zealand (Beentjes & Carbines 2005, Wade 2020, Kolodzey et al. 2023). The distribution of large kelps has been negatively impacted by marine heatwave events and sedimentation from land-based runoff along parts of the South Island (See bottom-up forcing in Table 13.3, Tait et al. 2021, Thomsen et al. 2019). In these same regions blue cod abundance has shown a strong negative correlation with environmental variables such as sedimentation and increasing sea surface temperature, though it is uncertain whether these variables are affecting blue cod directly or indirectly through effects on food and habitat (Brough et al. 2023). Juvenile blue cod in particular are likely to be negatively impacted by the effects of sedimentation as they are known to inhabit shallow inshore reef margins associated with turfing algae and sponges that are exposed to land-based inputs (while adults are typically associated with deep reefs) (Brough et al. 2023).

A decrease in macroalgal habitat complexity could reduce the abundance of blue cod by suppressing the availability of key prey species (Wade 2020). A New Zealand case study, using chemical tracers of diet for five fished species (blue cod (*Parapercis colias*), hapuku (*Polyprion oxgeneios*), barracouta (*Thyrsites atun*), red cod (*Pseudophycis bachus*) and ling (*Genypterus blacodes*), suggests that the historical decline of kelp forests has resulted in a less connected and more simplistic foodweb, and an increased energetic cost (reliance of fished stocks on higher trophic level foods) of fisheries productivity (Wing et al. 2022a).

Fishing and ecosystem effects are likely to have a cumulative impact on Blue Cod stocks. Kolodzey et al. (2023) compared the length frequency distributions, growth, stomach contents, and isotopic values ($\delta^{13}C$ and $\delta^{15}N$) of blue cod subpopulations at healthy biogenic reef sites and degraded sites impacted by dredging and high turbidity from suspended sediment in the Marlborough Sounds and Tasman Bay. Blue cod were significantly smaller and occupied smaller isotopic niches at the sites degraded by sediment and bottom fishing compared to healthy reef sites, which was attributed to the impacts of these factors on prey availability. Adult blue cod also have a small home range and are protogynous hermaphrodites[5], with some fish changing from female to male later in life. This life history strategy makes a population's sex ratios particularly vulnerable to fishing impacts (e.g. large males are more likely to be fished). Overfishing may then cause females to change sex, resulting in more small male fish and fewer large females. This leads to a less productive population, as observed in the Marlborough Sounds (Kolodzey & Wing 2022). Thus, while bottom-up forces affect the ecosystem structure and function required to support healthy blue cod populations, it is the combination of both bottom-up environmental factors and the top-down effects of fishing that cumulatively impact their role as an important predator in New Zealand's coastal ecosystems.

Bottom-up environmental impacts on long-spined sea urchins

Figure 13.4 *Centrostephanus rodgersii* Copyright: John Turnbull www.marineexplorer.org

In New Zealand, the population structure of the long-spined sea urchin (*Centrostephanus rodgersii,* Figure 13.4) appears to be changing under warming conditions (Doheny et al. 2023). Like kina (*Evechinus chloroticus*), the long-spined sea urchin is an echinoderm capable of contributing to trophic cascades leading to sea urchin barrens (Byrne & Andrew 2020, Davis et al. 2023). Long-spined sea urchins can create more persistent sea urchin barrens than kina because they occupy a greater depth range (Perkins et al. 2015) are more mobile and omnivorous, and potentially less vulnerable to predation (Mayer 2024, Balemi & Shears 2023). Packhorse rock lobster (*Sagmariasus verreauxi*) and spiny lobster (*Jasus edwardsii*) are the only known predators of Centrostephanus in New Zealand (Balemi & Shears 2023). While the distribution of long-spined urchins in eastern Australia has been expanding southwards as a result of increasing sea surface temperature (SST) (Byrne & Andrew 2020, Davis et al. 2023), the same extent of range expansion has not yet been observed in New Zealand populations. However, changes in the population and key biological characteristics indicate that this species has the potential to increase in range and abundance, with negative implications for the inshore ecosystems. Pecorino et al. (2013, 2014) found that further ocean warming in New Zealand could improve conditions for larval and embryonic development of long-spined sea urchins, increasing the possibility of marked range expansion. This expansion is likely to

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[5] Animals that are born female and at some point later in their lifespan change sex to male

result in reduced resilience of coastal marine ecosystems to fishing effects and demand more active management to prevent catastrophic phase shifts (Ling et al. 2009).

In 2023 Australian fisheries management authorities developed the National Centro Task Force to allow for collaboration on management activities and prioritisation of environmental health in response to the spread of long-spined urchin barrens (Keane et al. 2024). The proliferation of long-spined sea urchins in Australia is largely associated with climate-driven changes in ocean currents promoting sustained ocean warming (favourable conditions for long-spined larvae) and kelp loss (Ling et al. 2008). The vulnerability of kelp beds to climate-driven phase shifts to urchin barrens is thought to be reduced by fishing of reef predators such as spiny lobsters (Ling et al. 2009). While spiny lobsters can feed on long-spined urchins, they do not appear to be effective at reducing the extent of urchin barrens in Tasmania as they prefer native species over the range-extending urchins (Smith et al. 2022) and are less effective predators on the larger urchins found in established barrens (Smith et al. 2023). The abalone industry, which was threatened by the expansion of sea urchin barrens, has invested heavily in Tasmania's response thus far by creating harvest subsidies for long-spined sea urchins, lobbying for political support and enabling the development of a commercial fishery for urchins. The Australian government is yet to allocate large-scale funds to support an ecosystem-based approach to addressing the spread of long-spined sea urchins, but steps to achieve this, including marine spatial planning approaches are outlined in the 5-year plan published in 2023 (National Centro Taskforce 2023).

13.4.2 CASE STUDY 2: TOP-DOWN FORCING AND SEA URCHIN BARRENS

Figure 13.5 Urchin barren versus healthy kelp covered reef in Northland, New Zealand. Copyright: Vince Kerr

Background on fishing-induced trophic cascades leading to sea urchin barrens

Fishing is thought to play a role in the development of sea urchin barrens in parts of New Zealand, and this relationship provides an example of top-down forcing resulting in a 'trophic cascade' (term defined in Section 13.2.2). Although multiple factors can cause kelp decline (including sedimentation and marine heatwaves), evidence from marine reserves in northeastern New Zealand suggests that fishing of apex reef predators is an important factor behind the proliferation of the grazing sea urchin/kina (*Evechinus chloroticus*), extensive kelp loss and the expansion of sea urchin barrens (Doheny et al. 2023). This relationship is based on observations of the concurrent recovery of kelp (*Ecklonia radiata*) and predators of sea urchins (including snapper, *Chrysophrus auratus* and spiny lobsters, *Jasus edwardsii*) inside long-term marine reserves near Leigh (Babcock et al. 1999, Shears & Babcock 2002, 2003, Leleu et al. 2012), and the positive effect of protection from fishing on the abundance of kelp (*Ecklonia radiata*) and predators inside seven marine reserves from the Kermadecs to the Bay of Plenty (Edgar et al. 2017).

Extent of sea urchin barrens and interacting effects on the ecosystem

Historical photography and local knowledge indicate urchin barrens were first observed in northeastern New Zealand around the 1960s with barrens becoming quite extensive by the 1970s, and some continued expansion since then (Booth 2017). An estimated 30% (range: 7 – 49%) of shallow reef area is impacted by sea urchin barrens from Tāwharanui Peninsula in the Hauraki Gulf to Maitai Bay near the tip of the Karikari Peninsula in Northland (Kerr et al. 2024) with even higher estimates of up to 90% in parts of the Bay of Islands (Booth 2017). There is no published evidence of bottom-up forces such as ocean warming affecting kelp forest extent in these areas yet, but it is a concern for the future (Cornwall et al. 2023). The degraded state of these reefs may make them less resilient to the impacts of climate change and is likely to be already impacting ecosystem function. Urchin barrens support a lower biodiversity due to the loss of ecosystem services, including provision of complex three-dimensional habitat and organic matter, provided by kelp forests (Rogers-Bennett & Catton 2019, Udy et al. 2019) (Figure 13.5). For example, the loss of kelp forests could reduce the capacity of the ecosystem to support healthy spiny lobster populations due to their potential reliance on these habitats across multiple life stages. Studies from Tasmania suggest kelp habitat may be critical to the settlement success of spiny lobster pueruli (post-larval spiny lobster), providing important settlement cues, food and refuge (Hinojosa et al. 2015, Hinojosa et al. 2018, Shelamoff et al. 2022). The same relationship has yet to be observed for pueruli in New Zealand (Stanley et al. 2015, Hesse et al. 2016) though kelp is an important food source and sheltering habitat for older life stages of spiny lobster (MacDiarmid et al. 2013). These observed and potential relationships across the lobster life cycle suggest that prioritising kelp recovery is important from both an ecosystem and fishery perspective and that these two areas are tightly linked.

Regional and temporal variability

Where and when kina barrens occur within any region depends on abiotic factors such as turbidity, exposure and bathymetry that may affect the suitability of habitat for settlement of kina and kelp (Shears et al. 2008, Kerr et al. 2024). Short-term biotic disturbances such as disease or toxic dinoflagellate blooms can cause temporal variability in sea urchin (Shears & Ross, 2010) and kelp dynamics (Babcock & Cole 1993) but long-term trends suggest top-down control of predators is the primary mechanism controlling urchin barrens in northeastern New Zealand (Peleg et al. 2023). Experimental work in the Hauraki Gulf in 2020–21 recorded rapid kelp recovery in areas where urchin barrens were cleared by divers, suggesting that kelp re-growth is unlikely to be limited by environmental/bottom-up factors in these areas (Miller et al. 2024). In 2024 similar research led by the University of Auckland is in progress in Queen Charlotte Sound, Marlborough. Once in place, urchin barrens dominated by large urchins are persistent, and recovery of macroalgal beds requires very low urchin abundance (Miller et al. 2022).

Outside of northeastern New Zealand, and especially in central areas like the Marlborough Sounds, environmental and climatic influences, species' demographics, and catchment-derived sedimentation may be equally or potentially more important than predator abundance in structuring kelp beds (Doheny et al. 2023, Schiel 2013, Wing et al. 2022b). More research is needed to better understand the prevalence of kina barrens and contributing factors in these regions. Observations of concurrent kelp and pāua recovery inside a marine reserve in Stewart Island between 1998 and 2014 suggest that fishing of pāua may result in kina barrens since pāua may help maintain community structure by displacing sea urchins from shallow water habitats (Wing et al. 2015).

Key Predators

Key predators involved in trophic cascades leading to kina barrens and their relative importance will vary regionally depending on their distribution. Predators that consume kina include spiny lobster, packhorse lobster (*Sagmariasus verreauxi*), large sea stars and several species of fish (e.g. snapper, red moki (*Cheilodactylus spectabilis*), blue cod (*Parapercis colias*), and banded wrasse (*Notolabrus fucicola*) (Doheney et al. 2023). Evidence from marine reserves in northeastern New Zealand suggest that increased abundances of predators (including snapper and spiny lobster) were associated with the re-establishment of kelp forests in former urchin barrens. The role of spiny lobsters as keystone predators is also reflected in the results of ecosystem models in New Zealand rocky reef ecosystems, using both qualitative (Beaumont et al. 2009) and quantitative frameworks to estimate historical food web structure (Pinkerton et al. 2008, Eddy et al. 2014, Pinkerton 2012). While sea urchins are not the preferred prey of spiny lobsters, which choose soft-sediment bivalves over urchins when provided the option, and consumption is likely to vary seasonally with lobster moulting stage and/or sea urchin roe quality, large lobsters are unique in their ability to pry large urchins from rocks and consume them via the urchins' unprotected mouthparts (Flood, 2021). Urchin predation by fish such as snapper is limited by mouth gape, so only large snapper can consume large sea urchins (Marinovich et al. 2023). Thus, larger urchins may be protected from predation by most predators other than large (>130mm carapace) lobster (Andrew & MacDiarmid 1991). The presence of both spiny lobster and snapper can also influence sea urchin behaviour. For example, increased presence of these predators inside marine reserves increases cryptic behaviour (such as hiding in crevices) by sea urchins (Spyksma et al. 2017) and sea urchins consume less kelp in the presence of lobsters (Curtis & Wing 2024). Thus, snapper and rock lobster potentially mitigate trophic cascades by consuming urchins and by reducing their consumption of kelp.

Although kina were historically the primary species found in sea urchin barrens in New Zealand, warming waters are causing an increase in barrens dominated by long-spined sea urchins (as described in 13.4.1).

Management approaches to address urchin barrens

Manual removal of kina from barrens can support rapid kelp regrowth in the short-term (Miller et al. 2022, Villouta et al. 2001), but the benefits are temporary and do not promote recovery of sea urchin predators (Miller et al. 2024). Potential management solutions to address the top-down effects of fishing on the ecosystem require a suite of approaches including managing kina predators to higher abundance, spatial controls on fishing, and targeted removal of sea urchins in priority areas (Bulmer et al. 2024, Miller et al. 2022, Doheny et al. 2023, Kerr et al. 2024). However, the relative effectiveness of these measures is difficult to quantify and will require monitoring over time. The biomass of predators required to reverse, mitigate, or prevent urchin barrens is currently unknown. Sustained recovery of kelp forests in Northland has only been observed in areas closed to all fishing, not in partial closures (Shears et al. 2006), suggesting that no-take areas, and the biomass of predators they support, are an important part of the solution (Nessia et al. 2024).

13.4.3 CASE STUDY 3: MIDDLE-OUT/WASP-WAIST FOOD WEBS ON THE CHATHAM RISE

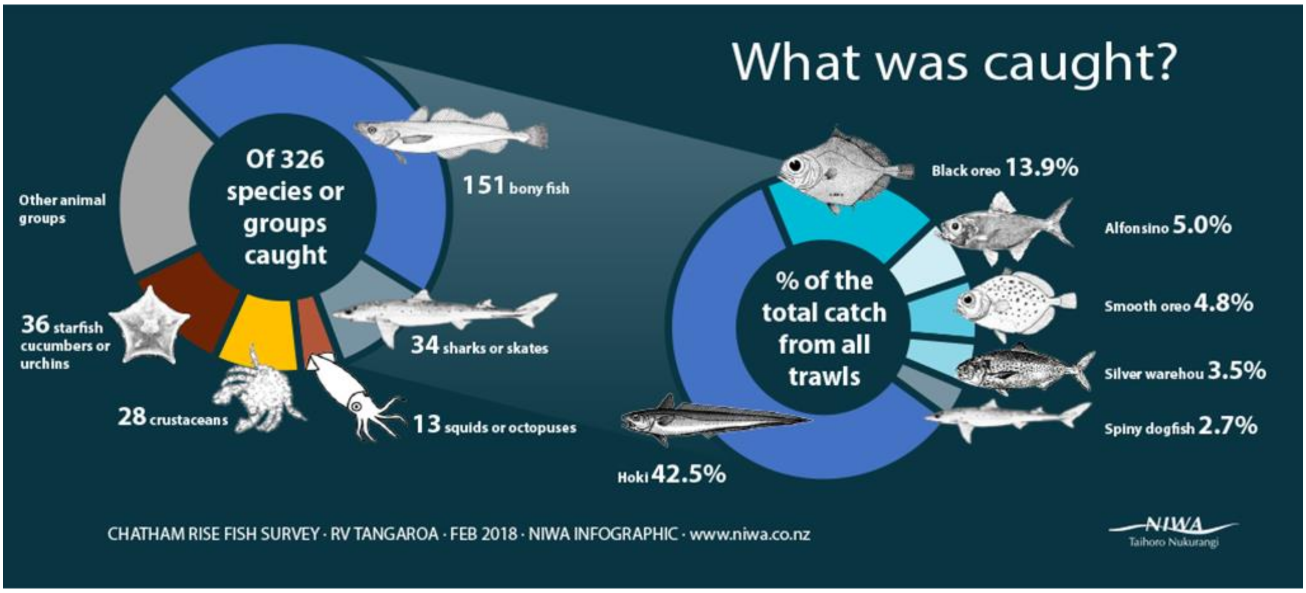

Figure 13.6 NIWA infographic illustrating the fish caught on fishery-independent Chatham Rise fish surveys (https://niwa.co.nz/news/niwa-expertise-contributes-healthy-hoki-fishery).

Research into deepwater ecosystems in the New Zealand EEZ is most advanced in the Chatham Rise region (Pinkerton 2013) which provides an example of a wasp-waist controlled ecosystem. Elevated primary production here is due to the convergence of subantarctic and subtropical water which is called the Subtropical Front (Bradford-Grieve et al. 1997, Boyd et al. 1999, Murphy et al. 2001, Sutton 2001). This front supports valuable deepwater fisheries, an unusually rich benthic ecosystem (Probert et al. 1996, McKnight & Probert 1997, Bowden 2011), and large seabird populations (Figure 13.7;Taylor 2000a, 2000b). Ecosystem modelling of the Chatham Rise food web began in 2006 (e.g. Pinkerton, 2013), the most recent version being McGregor et al. (2019), using the Atlantis ecosystem model. McGregor et al. (2019) calculated 'keystoneness'[6] (Libralato et al. 2006) and responsiveness to perturbations as measures of trophic importance in the Chatham Rise Ecosystem (CRE). The model used 53 functional groups to model biological processes across various fish, cetaceans, other mammals, seabirds, invertebrates, zoo/phytoplankton, and bacteria. These groups were modelled with biotic and abiotic variables such as depth, sea surface temperature, salinity, nutrient availability, predation effects, and fishing pressure, over a period of 1900–2015. Fish groups that ranked highest for keystoneness were; hoki (*Macruronus novaezelandiae*), orange roughy (*Hoplostethus atlanticus*), shallow benthic fish (primarily oblique banded rattail), and hake (*Merluccius merluccius*). Small to medium pelagic fish (primarily lanternfish (myctophids) and barracouta (*Thyrsites atun*) respectively) ranked next for keystoneness. These results suggest some degree of middle-out control in the system, although the number and functional diversity of these groups is higher than in other wasp-waist systems.

Other work on the Chatham Rise food web investigated change in food web structure over time using data from Chatham Rise trawl surveys between 1992 and 2007 (Tuck et al. 2009). See Figure 13.6 for a summary of key species caught in research trawl surveys. This work found evidence of increasing evenness (reducing diversity) but no evidence that species were being lost from the food web (Tuck et al. 2009). Over the sampling period, fish caught in research trawls became smaller and the mean trophic level index decreased (Pinkerton 2013), though it has subsequently increased. The proportion of piscivorous fish and of true demersal (rather than bentho-pelagic) species declined while threatened species (several species of sharks and rays) and 'low-resilience 'species, such as dogfish and rays increased relative to other species on the Chatham Rise (Tuck et al. 2009, O'Driscoll et al. 2011). This may be a result of a combination of a lack of incentive to catch these species by the fishing fleet and an increase in offal and discards that

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[6] Keystoneness is defined as the impact of a species on different elements of an ecosystem resulting from a small change to the biomass of the species and its total biomass.

benefit demersal scavengers. There were also changes in the spatial distribution of fish species. MPI project ZBD2004/02 (Dunn et al. 2009, Horn & Dunn 2010) found evidence that fish (primarily myctophids such as lanternfish) were increasing in importance in the diet of hoki, but not of hake or ling.

13.5 METHODS FOR MEASURING THE SCALE AND SIGNIFICANCE OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

13.5.1 MEASURING THE SCALE OF ECOSYSTEM EFFECTS

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 (outside of New Zealand's EEZ) 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 (for example southern bluefin tuna), but others remain in a small area all their lives (e.g., on a reef). Marine ecosystems should therefore 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, Stephenson et al. 2018, 2020); (c) relating community composition to environmental variables (e.g., generalised dissimilarity analysis; Leathwick et al. 2012, Stephenson et al. 2022) and using these relationships to extrapolate spatially; (d) using an image based approach to identify vulnerable marine ecosystems (Gros et al. 2023); and (e) analysing environmental DNA (eDNA) data to assess changes in the proportional abundance of taxa and to analyse the linkages between organisms (Djurhuus et al. 2020).

13.5.2 MEASURING THE SIGNIFICANCE OF ECOSYSTEM EFFECTS

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 (Brose et al. 2005, Pascual & Dunne 2006, Brander 2010, Jennings & Brander 2010). Ecosystems are complex adaptive systems (Levin 1998, 1999); that 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 temporal and spatial scales and the wide variety of potential stressors makes it difficult at present to forecast the response of ecosystems to change or specify tipping-points to guide management.

Several multispecies or ecosystem models have been developed that can be used to investigate the potential for trophic and ecosystem-level effects (Plagányi 2007, Plagányi et al. 2014). These include Ecopath with EcoSim (EwE; Watari et al. 2019, Craig & Link 2023), mass-balance (Pinkerton, 2010, Pinkerton 2013, Pinkerton et al., 2010, 2015a, 2023), Atlantis (Ortega-Cisneros et al. 2017, McGregor et al. 2019), OSMOSE (Grüss et al. 2015, Xing et al. 2021), Bayesian Belief Network models (Spiers et al. 2016, Trifonova et al. 2021) 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 (Fulton et al. 2005, Smith et al. 2011, Craig & Link 2023, Pinkerton et al. 2023). 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). Knowledge of food web structure (Table 13.2) has also been used to evaluate responses to fishing and other changes as an alternative to dynamic ecosystem models (Ulanowicz & Puccia 1990, Libralato et al. 2006, Pinkerton &

Bradford-Grieve 2014, Pinkerton et al. 2023). Progress continues to be made to simplify methods for evaluating fishing impacts on the ecosystem and standardise approaches to advancing ecosystem-based fisheries management, including the development of ecosystem reference points (Fulton & Sainsbury 2024).

13.5.3 EXAMPLES FROM NEW ZEALAND

Several pieces of work have contributed to our understanding of the spatial distribution of marine ecosystems in New Zealand (e.g., Leathwick et al. 2006, 2012, Stephenson et al. 2018, 2020, 2022). Several approaches have been developed in New Zealand to identify or describe the estimated 62 distinct marine habitat types (Spalding et al. 2007, MacDiarmid et al. 2012) including a marine habitat classification index developed by New Zealand's Department of Conservation and MPI (Department of Conservation and Ministry of Fisheries 2011) and more recently the New Zealand Seafloor Community Classification (Stephenson et al. 2022) and the New Zealand Seafloor Bioregionalisation (Stephenson et al. 2023), characterising environmental conditions.

Despite New Zealand's small land mass, the coastline covers a wide latitudinal range with diverse geography and is exposed to widely varying environmental conditions. Thus, trophic relationships and ecosystem-level changes can vary in different parts of the country. For example, although marine heatwaves have become stronger, longer and more frequent overall in New Zealand, seasonal trends vary both within and between regions, with implications for the resilience of different ecosystems to climate change (Montie et al. 2024). These impacts can vary at quite localised scales, with kelp loss and growth stunting in pāua attributed to heat stress varying between reefs along a short stretch of coastline in the Chatham Islands (Van Nguyen et al. 2023).

Multiple ecosystem models have been developed to describe significance of fishing effects on ecosystems in New Zealand. For example, one long-term study characterised changes to the ecosystem of the Hauraki Gulf region using five balanced, quantitative models of the food web covering periods from the present day back to pre-human settlement (Pinkerton et al. 2015b). 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, with strong top-down ecosystem control. With the loss of seals/sea lions from the Hauraki Gulf ecosystem and the decline in cetaceans, models suggest their trophic importance was drastically reduced, potentially impacting populations of top predators such as white sharks (*Carcharodon carcharias*) and orca (*Orcinus orca*) and their prey (MacDiarmid et al. 2016). The trophic importance of other predators (spiny lobsters and sharks especially) in the models of the Hauraki Gulf ecosystem also reduced over time as a result of human harvesting suggesting a transition to a more bottom-up controlled system.

Other ecosystem models have described the historical impacts of predators on rocky reefs in the Hauraki Gulf (MacDiarmid et al. (2016), the impacts of fishing on the Chatham Rise (McGregor et al. 2019), the impacts of fishing for Antarctic toothfish (*Dissostichus mysoni*) on the ecosystem of the Ross Sea, Antarctica (Pinkerton et al. 2010, Pinkerton & Bradford-Grieve 2014), and the potential for the toothfish fishery to indirectly affect the abundance of Adelie penguins (Pinkerton et al. 2016). There are significant uncertainties in the parameters used in these models and the authors note that the results represent one of many possible solutions, and should be considered as working hypotheses. Nonetheless these models provide a valuable approach to collate the best available knowledge on large scale changes to ecosystems over time.

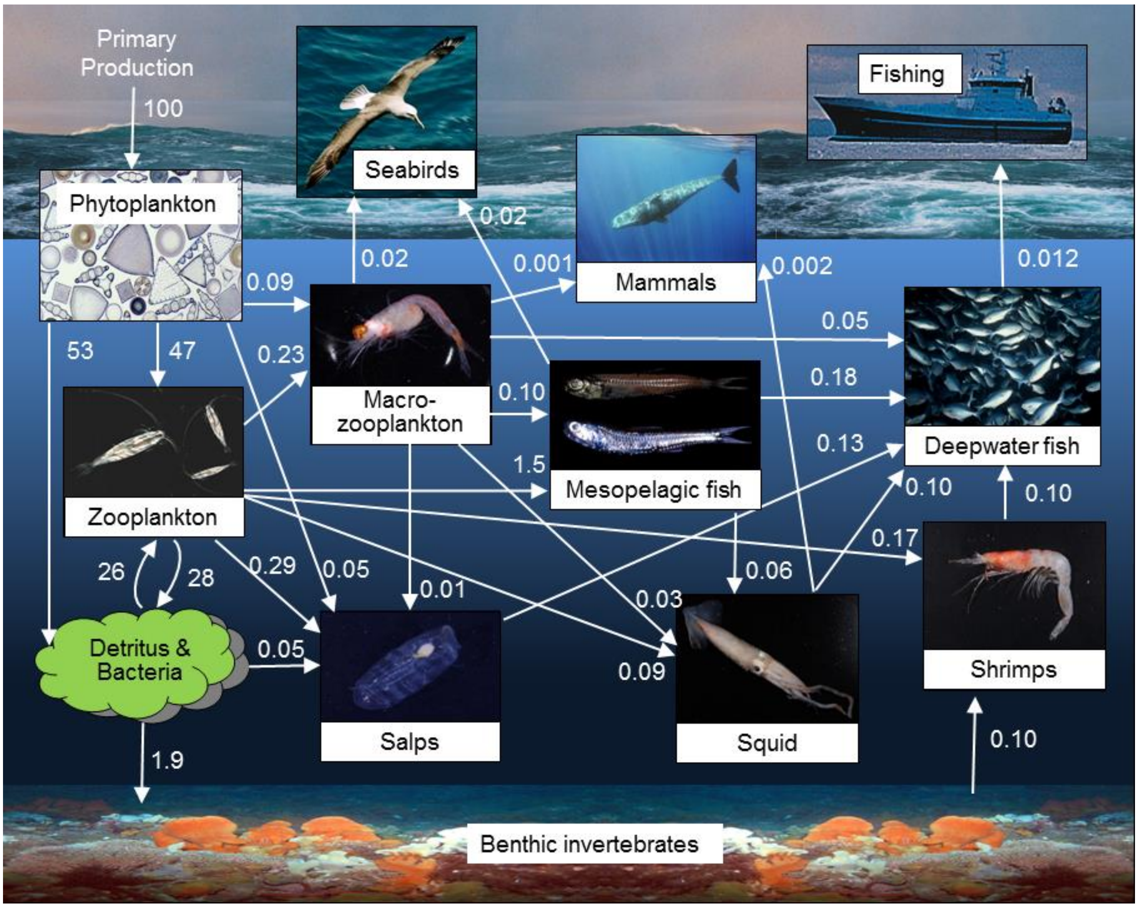

Figure 13.7: Simplified trophic model of the Chatham Rise, New Zealand (based on Pinkerton 2013). The growth of phytoplankton generates organic matter that is the fuel for the marine ecosystem. Figures show the annual flow of energy through a unit area of the food web normalised to a net primary productivity (NPP) of100, based on an equilibrium mass-balance model (similar to Ecopath).

13.6 METHODS FOR DETECTING TROPHIC AND ECOSYSTEM EFFECTS

Detecting trophic and ecosystem-level effects can be difficult as they often occur over broad temporal or spatial scales. This is particularly prevalent in the marine environment, where ecosystems and ecological thresholds can be difficult to define (Bland et al. 2018). At regional scales the easiest way to measure the effects of species on an ecosystem is to look at comparable habitats where the species is present and absent, either through experimental manipulation or natural experiments where species appear or disappear naturally over time or space.

Over broader scales ecosystem indicators can offer valuable insights. There is a 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, McQuatters-Gollop et al. 2015). Given the multi-trophic nature of ecosystem-level effects and the interest in advancing approaches to ecosystem-based fisheries management (EBFM) (Rogers & Greenaway 2005, Tam et al. 2017, Thompson et al. 2019), indicators are needed that span the ecosystem, including primary producers, the microbial system, middle trophic levels, fish communities, the benthic community, and top predators. EBFM is an integrated management approach that includes consideration of both social and ecological components. A summary of some recommended indicators can be found in Appendix 1.

There has been much work in New Zealand on developing indicators of the marine environment. Fisheries New Zealand have carried out projects looking at indicators and time series, including oceanographic/climate variables (Hurst et al.

2012, Pinkerton et al. 2015a), primary production (D'Archino et al. 2019, 2021), 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), national environmental reporting (Pinkerton 2007, Pinkerton 2014, Department of Conservation 2022) and a research project in the Hauraki Gulf funded by the Sustainable Seas National Science Challenge (https://www.sustainableseaschallenge.co.nz/our-research/ebfm-in-the-hauraki-gulf/). Existing information that can be used to detect trophic and ecosystem effects in New Zealand is summarised below.

13.6.1.1 MARINE PRIMARY PRODUCTION

Ocean colour satellite data, measuring concentrations of chlorophyll-a, has been used in New Zealand to investigate spatial and seasonal patterns in phytoplankton abundance and net primary production (NPP) (Murphy et al. 2001, Pinkerton 2007, Jiang et al. 2017, Pinkerton et al. 2019, 2024a). While there are other methods of estimating NPP from satellite data (Behrenfeld et al. 2005, Westberry et al. 2008), the concentration of chlorophyll-a (chl-a) is a more accurate and preferred metric for the purpose of monitoring change in primary production over time (Pinkerton et al. 2014, 2019, 2024a). Since 2002, mean concentrations of chlorophyll-a in the EEZ have varied spatially and temporally, with changes likely to be related to oceanographic cycles such as the Interdecadal Pacific Oscillation index and the Southern Oscillation Index, as well as to long-term climate change (evidence of surface warming since 1981). In the most recent update (Pinkerton et al., 2024a), at the ocean scale, increasing trends in chl-a were found in the Subtropical Front including a moderate increasing trend over the Chatham Rise (+1.0% of the median per decade), and a greater relative change in Subantarctic water to the south (+6.4% per decade). Conversely, a declining trend (-4.4% per decade) was identified in Subtropical waters located northeast of New Zealand. These findings generally align with previous analyses and our understanding of factors influencing primary productivity in the waters surrounding New Zealand. However, an unexpected and rapid decline in chl-a in all New Zealand ocean regions was observed since mid-2019 which has continued to late-2023. The rapid decline in chl-a in the New Zealand ocean domain may have ecological implications, including for New Zealand deepwater fisheries in the coming years.

At the New Zealand coastal scale, territorial waters have warmed over the last 40 years, and coastal scale marine heatwave events have generally become more frequent and longer-lasting (Pinkerton et al. 2024a). Since 2002, satellite observations have identified increases in chl-a around the coasts of mid-lower North Island and all South Island, particularly during winter. In contrast, coastal regions along the west coast of Northland and northeast New Zealand shelf including the Hauraki Gulf, Coromandel, and Bay of Plenty displayed decreasing trends in chl-a (productivity). Lower-than-normal chl-a was observed along the west coast of South Island during the MHW of December 2017 to March 2018 and December 2022 to January 2023 (Pinkerton et al. 2024a).

Satellite imagery data was analysed by Tait et al. (2021) to assess changes in surface canopy abundance of giant kelp (*Macrocystis pyrifera*) along the southeastern coastline of New Zealand. Sixty months of imagery was analysed for the period between December 2015 to December 2020, including the marine heatwave in 2017/2018. The approach computes vegetation indices based on measurements of near infrared electromagnetic radiation. The imagery detects vegetation not obscured by water, giving only the measurements of the floating macroalgal canopy and no indication of the full *Macrocystis* community. However, surface cover is still a key indicator of health in giant kelp populations and therefore remote sensed time series have great potential for identifying responses of kelp communities (important primary producers) to environmental stressors (Cavanaugh et al. 2011, Tait et al. 2021). Satellite imagery and aerial photography has also been used to map the spatial extent of kina barrens, which can highlight areas where kelp loss has occurred (Kerr et al. 2024, Dartnall 2022) and these methods will be applied to Fisheries New Zealand funded research project ZBD2023-03 in 2024/25.

13.6.1.2 LOWER FOOD WEB (MICROBIAL SYSTEM)

Changes to primary production are likely to translate to less food available for higher trophic levels. Virtually all wild-caught seafood in New Zealand is 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 has 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 or the efficiency of marine trophic transfer.

13.6.1.3 MIDDLE TROPHIC LEVELS

Middle trophic level organisms in the New Zealand ocean are diverse, from zooplankton, copepods, and krill to larger organisms such as myctophids (lanternfish) and forage fishes (e.g., anchovy, pilchards, and sprat). Although they form the basis of the diet of many commercially important New Zealand fish species (Dunn et al. 2009), the basic abundance, distribution, and ecology of key middle-trophic level groups such as myctophids and hyperbenthic arthropods (prawns and shrimps) are generally poorly known (Escobar-Flores et al. 2018). 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 annually since then on transits of a commercial fishing vessel to the Southern Ocean (Pinkerton et al., 2024b) in collaboration between NIWA and a commercial fishing company (Sanford Ltd), funded by MPI. The transits extend from Oamaru (approximately 45°S) to the Ross Sea; approximately 1200 km of this transect are in the subantarctic New Zealand EEZ (Robinson et al. 2021). The overall objective of the project is to understand spatial, seasonal, and long-term changes to zooplankton in New Zealand's southern EEZ and on the transit to the Ross Sea (Robinson et al. 2021). The latest analysis found a decreasing trend in phytoplankton, primary production, and total zooplankton abundance in the Ross Sea sector between 2008 and 2023, with an average rate for zooplankton of -1.9% of the mean per year (Pinkerton et al., 2024b). Trajectories of change differed between taxonomic groups of zooplankton, with some increasing. In contrast, phytoplankton, primary production and modelled environmental suitability for zooplankton generally increased in other parts of the Southern Ocean (+0.3% per year for zooplankton environmental suitability).

Other methods to estimate abundances of mid-trophic level organisms (MTLO) include multifrequency backscatter acoustic data (Escobar-Flores et al. 2022). From 2008–2014, acoustic data was analysed on 28 transects between New Zealand and the Ross Sea to identify changes in spatial and temporal patterns of MTLOs. While acoustic backscatter can be used to estimate changes in relative abundances of MTLOs over time, it is limited in its ability to estimate total biomass (Kloser et al. 2016, Pinkerton et al. 2023).The same method has been used to derive indices of abundance of mesopelagic fish and invertebrates on the Chatham Rise (McClatchie & Dunford 2003, O'Driscoll et al. 2009) and could provide valuable time series of middle trophic level species in the Hauraki Gulf and subantarctic plateau (Pinkerton et al. 2023).

13.6.1.4 DEMERSAL FISH COMMUNITIES

There are three series of scientific trawl surveys 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 were done in the Hauraki Gulf region between 1980 and 2000 and 2019 to present day (Parsons & Bian 2022). 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.

13.6.1.5 TOP PREDATORS

Top predators in New Zealand can also act as 'sentinel' species. Changes in their ecological behaviour such as distribution or diet preferences can indicate changes in trophic interactions and or environmental effects (Peters et al. 2022, Ogilvy et al. 2022). Sperm whales (*Physter macrocephalus*) and blue whales (*Balaenoptera musculus*) can act as excellent indicator species due to their long lifespan and high sensitivity to environmental shifts in their prey distribution and abundance (Hazen et al. 2019). For example, blue whale migrations are easy to observe, and their foraging patterns can offer predictions of trends in primary productivity (krill). Models based on three IPCC (Intergovernmental Panel on Climate Change) climate scenarios suggest that distributions of these species will shift southwards within the New Zealand EEZ by 2100 (Peters et al. 2022). Scenarios were based on Representative Concentration Pathways (RCP) 2.6, 4.5, and 8.5 (Pachauri et al. 2014). These pathways represent varying climate change scenarios in 2100, depending on the amount of greenhouse gases emitted in future, with 2.6 the least (~400 ppm CO_2e) and 8.5 the most (~1200 ppm CO_2e). The most severe scenario tested (RCP 8.5) generated 61% and 42% suitable habitat loss for sperm and blue whales respectively, mostly from New Zealand's northern waters.

The Māui dolphin (*Cephalorhynchus hectorii maui*) is a critically endangered cetacean that is endemic to New Zealand. Understanding its diet and foraging ecology could provide insights into its trophic relationships, habitat selection, and the health of the wider ecosystem (Derville et al. 2016). Decreases in stable isotopes $\delta^{13}C$ and $\delta^{15}N$, gathered from skin samples of Māui's dolphins between 1993 to 2021 suggests substantial changes to diet since the implementation of the Marine Mammal Sanctuary along the west coast of the North Island (Ogilvy et al. 2022). There is some concern that ENSO events are negatively impacting key dolphin prey such as juvenile red cod (Miller et al. 2013, Beentjes & Renwick 2001) and further monitoring of Māui's dolphin diet may provide a useful indication of climate change impacts to the food web.

13.7 DISCUSSION

Marine ecosystems are complex, show uneven responses to disturbance and are subject to a wide range of impacts, including fishing, climate variability and change, coastal runoff and eutrophication, habitat change, invasive species and potentially plastics. Any activities that change the composition of species in the ecosystem (in terms of size, functional group, ecosystem role, and diversity) can affect other groups in the ecosystem through trophic links and other relationships. There are thresholds that must be met before a change is observed, with more degraded systems having lower thresholds or resilience to impacts (Casini et al. 2009). 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, Bakun & Weeks 2006, Valdes et al. 2009, Worm et al. 2009, Lindegren et al. 2010, Rogers-Bennet & Catton 2019). Understanding and quantifying 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 (Lindegren et al. 2016, Mumby et al. 2022) and 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, fishing is likely to contribute cumulatively alongside cross-sector and environmental impacts to make ecosystems less resilient to variability and change in climate/oceanographic forcing (Winder & Schindler 2004, Kirby et al. 2008, 2009). 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 identify at what point fishing or other pressures 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 (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 Ocean Research/Intergovernmental Oceanographic Commission working group; Cury & Christensen 2005) suggest that harvesting many species in an ecosystem at B_{MSY}[7] 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, Edgar et al. 2024).

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 acknowledging the potentially synergistic effects of fishing and climate change (CBD 2009, Perry et al. 2010, Rice & Garcia 2011). This process of considering ecosystem level effects and interactions in fisheries management is commonly referred to as 'Ecosystem-based fisheries management' (EBFM; Townsend et al. 2019, Howell et al. 2021).

Ecosystem based fisheries management (or name variations thereof) is being increasingly implemented and used by governments around the world. Early fundamental work by the National Ocean and Atmospheric Association (NOAA) Fisheries (Marasco et al. 2007) outlined pragmatic approaches to ecosystem-based fishery management that can still be used today:

- incorporating a broader array of societal goals and uses for ecosystem products and services within a multiple use multiple stressor 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.

Building on these initial principles, NOAA Fisheries have developed an EBFM Policy and Roadmap that provides guidelines for the implementation of EBFM (National Marine Fisheries Service 2016). This document supports eight individual implementation plans to address the specific ecosystem needs and environmental conditions for regions around the United States. A full breakdown of global EBFM implementation is beyond the scope of this specific chapter. However, an EBFM approach is required to address trophic cascades and other ecosystem effects in fisheries management decisions.

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[7] The biomass that allows the maximum sustainable yield to be taken.

There are differing opinions on the role of no-take reserves or marine protected areas (MPAs) in guarding against trophic and ecosystem-level effects. A full review of the value of MPAs in this regard is beyond the scope of this chapter. 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 others believe MPAs are too few and too small to have any value in this regard (Kaiser 2005, Mora et al. 2006, Ovando et al. 2021, LaScala-Gruenewald et al. 2021). Viable no-take marine reserves that are large enough that most ecological processes are able to operate within them and are of sufficient size to encompass the home ranges of species or species groups that are the target of protection 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, Babcock et al. 2010, Hanns et al. 2022).

Specific recommendations from Marasco et al. (2007) in relation to EBFM in US fisheries are relevant to recent research initiatives in New Zealand. In 2022, the Sustainable Seas National Science Challenge (SSNSC) initiated an ‘Improved decision making using an ecosystem-based management (EBM) approach” project that aims to deliver practical roadmaps for undertaking EBFM in New Zealand. The SSNSC also brought together research and provided input into regulation, policy and legislative reviews (Macpherson & Jorgensen 2024). Fisheries New Zealand commissioned further work in 2024 to advance an EBFM approach and incorporate ecosystem impacts into fisheries management decisions. These include an EBFM case study, and computer simulations to investigate the implications of incorporating ecosystem considerations into management targets and limits for fish stocks.

Addressing the issue of kina barrens has challenged New Zealand to advance the current ecosystem approach to fisheries management. A High Court decision in 2022 for the management of the CRA 1 spiny lobster stock in the Northland region confirmed that section 9 of the Fisheries Act 1996 provides an environmental bottom line requiring Fisheries New Zealand to incorporate ecosystem level issues, such as fishing-induced trophic cascades, into fisheries management decisions. Fisheries New Zealand held a National Kina Barrens workshop in Wellington in March 2023 as part of a research project on fishing-induced trophic cascades. This workshop provided a forum to share the most relevant science on the issue of trophic cascades leading to sea urchin barrens in New Zealand and developed a list of critical science and information required to support an EBFM approach (Doheny et al. 2023). Discussions with iwi and stakeholders to address the issue highlighted the need to incorporate mātauranga Māori, have harvest control rules for kina and predators and spatial planning. Outputs from this workshop are being progressed through science funding rounds and informing next steps for management.

Notwithstanding this progress, most New Zealand stocks are currently managed on a single-stock basis at close to B_{MSY} (Fisheries New Zealand 2024b) irrespective of their role in the ecosystem. Fishing close to B_{MSY} and in particular using bottom trawling (which impacts on benthic ecosystem function; Pusceddu et al. 2014) can 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 (Durante et al. 2021) 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 shifts which may or may not be reversible (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 be high value in maintaining regular fisheries-independent 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 the New Zealand Subantarctic Plateau.

13.8 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 fish stocks.

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, especially using bottom impacting gear such as trawling, can make ecosystems less resilient and more sensitive to the effects of environmental variability and change.

3. The issue of kina barrens provides a valuable example of top-down forcing on ecosystems in New Zealand. Research based on observations from marine protected areas suggests fishing of sea urchin predators is causing and/or maintaining sea urchin barrens in north-east New Zealand (Doheny et al. 2023). However, the role of fishing and the extent of kina barrens appears to vary around the country depending on local physical and biological variables and the extent of different impacts on the ecosystem (Wing et al. 2022b). Further research is needed to understand the extent of kina barrens and contributing factors in other regions.

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 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. Time series monitoring of fish communities and middle trophic level species (meso/macro-zooplankton, mesopelagics, and hyperbenthics) are crucial for understanding and monitoring for trophic and ecosystem-level effects. However, there is a paucity of long-term monitoring information for fish communities within the New Zealand EEZ.

6. Approaches to addressing ecosystem effects on fisheries requires a combination of traditional measures (quota allocation, catch settings) alongside strategic fishing closures, selective fishing gear and economic incentives. In New Zealand, linking these approaches depends on collaboration between Māori, fisheries scientists and managers, biologists and stakeholders to use the best available data to manage coastal ecosystems sustainably. Successful implementation will require co-developing approaches that incorporate mātauranga Māori (traditional knowledge) and principals of Whanaungatanga (kinship) through equitable partnerships.

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13.10 APPENDIX 1 – METHODS FOR DETECTING TROPHIC AND ECOSYSTEM EFFECTS

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[8] primary production (NPP) by phytoplankton growth in the ocean (Laufkötter et al. 2015, Le Grix et al. 2022). 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 (Roxy et al. 2016, Capuzzo et al. 2018).

LOWER FOOD WEB (MICROBIAL SYSTEM)

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[8] 'Net' means after allowing for phytoplankton respiration.

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 yield, 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 a 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.

MIDDLE TROPHIC LEVELS

Small mesopelagic[9] and hyperbenthic[10] 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 (Escobar-Flores et al. 2018, 2020). These 'middle trophic level' organisms are diverse and include hard-bodied crustaceans (such as copepods, euphausiids, amphipods, prawns and shrimps), 'jellies' (such as jellyfish and salps), cephalopods (squids and octopods), and a range of small fishes (including juveniles of larger species) living in the water column (especially myctophids or lanternfishes) or near the seabed. These species are likely to be affected both by fishing, which may reduce top-down predation control, and by climate-driven changes in lower trophic food web components (Frank et al. 2007, Richardson 2008). Middle trophic level species have a key role in ocean ecology (e.g., Banse 1995, Marine Zooplankton Colloquium 2 2001, Smetacek et al. 2004, Pinkerton 2013). Studying these middle trophic level organisms is challenging. They are typically diverse, with varied and complex life histories, can be hard to capture, and have abundances that vary over a wide range of space and time scales. Consequently, the factors that affect their dynamics are generally poorly understood. Two methods have been used for monitoring middle trophic levels. First, long time series of measurements of the zooplankton community by the Continuous Plankton Recorder (CPR) has demonstrated change in marine ecosystem (Mc Quatters-Gollop et al. 2015, Pinkerton et al. 2020, Suter et al. 2021), 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 (Trenkel & Berger 2013, Sakinan & Gücü 2017, Escobar-Flores et al. 2018, Mouget et al. 2020). However, this methodology requires more specialized knowledge and equipment than the CPR.

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, Durante et al. 2020). It was 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). MTI is scored between a 1 (primary producers) to 5 (apex predators). Gradual declines in trophic levels have been observed in many finfish fisheries around the world since industrialised fishing began (Pauly et al. 1998a, Christensen et al. 2003, Christensen et al. 2014). They vary regionally around the world based on targeted species, fishing intensity, and equipment used (, Adebola & de Mutsert 2019, Durante et al. 2020). These declines are generally 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' (Pauly et al. 1998a, Schiller et al. 2015). Essington et al. (2006) noted that 'fishing through the food web', where higher

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[9] 'Mesopelagic': inhabiting the intermediate depths of the sea, between about 200 and 1000 m down.

[10] 'Hyperbenthic': ecologically associated with the seabed but living for some time in the lower water column.

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, Durante et al. 2020). As such, MTI based on scientific surveys is likely to be a better indicator of change in fish communities (Branch et al. 2010, Durante et al. 2020). While the use of MTI to detect trophic shifts is widely used, modelling by Bourdaud et al. (2016) found that two new indicators, high trophic level indicator (HTI) and apex predator indicator (API) may be more suitable. The HTI is the ratio of the biomass of apex predators (TL>4) to the total biomass of consumers (TL>2) present in the ecosystem, and the API is the ratio of apex predators' biomass to the biomass of all groups whose trophic level is greater than 3.25 (Moullec et al. 2017). Trophic level cut-offs for both HTI and API were chosen to 'disentangle' the environmental effects that may disproportionately influence lower trophic organisms, from the effects of fishing.

- **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[11] 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[12] 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 (Eigaard et al. 2016, Tu et al. 2018). Several size-based metrics have been used to detect trophic and ecosystem-level changes (e.g., Collie et al. 2016, Lappalainen et al. 2016, Eigaard et al. 2016, Coll et al. 2008). Size-based indicators can be applied at a species or community level. Applied to a given species,

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[11] Those that produce relatively low numbers of offspring, typically growing more slowly and maturing later.

[12] Those that produce high numbers of offspring, typically growing faster and maturing sooner.

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 (Collie et al. 2016, Lappalainen et al. 2016).

- **Spatial distributions**: Fishing and climate 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).

TOP PREDATORS

Top predators (upper trophic level consumers) can be used in two ways as indicators of the state of marine ecosystems. First, one Organisation of Economic Co-operation and Development (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, 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. Top predators can also be indicators of ecosystem health, through the bioaccumulation of toxins and chemicals passed upwards through trophic levels. Halogenated organic compounds (HOCs) were found in five selected marine mammal species off the Southern California Bight. Cetacean species (long-beaked common dolphin, short-beaked common dolphin, and Risso’s dolphin) averaged 128 HOCs, whereas pinnipeds (California sea lion and Pacific harbour seal) averaged 47 HOCs (Cossaboon et al. 2019). This demonstrates that certain chemicals, even while they’re now banned, may still exist in the environment. 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).