Fisheries New Zealand
Tini a Tangaroa

# Distribution and potential causes of milky fleshed snapper in SNA 1

New Zealand Fisheries Assessment Report 2024/25

K.S. Johnson, J. Gadd, R. Bian, B. Noll,
M.H. Pinkerton, R. Taylor, B. Madden, D.M. Parsons

ISSN 1179-5352 (online)
ISBN 978-1-991285-72-0 (online)

**June 2024**

Te Kāwanatanga o Aotearoa
New Zealand Government

## Disclaimer



Requests for further copies should be directed to:

Fisheries Science Editor
Fisheries New Zealand
Ministry for Primary Industries
PO Box 2526
Wellington 6140
NEW ZEALAND

Email: Fisheries-Science.Editor@mpi.govt.nz
Telephone: 0800 00 83 33

This publication is also available on the Ministry for Primary Industries websites at:
http://www.mpi.govt.nz/news-and-resources/publications
http://fs.fish.govt.nz go to Document library/Research reports



Please cite this report as:

Johnson, K.S.; Gadd, J.; Bian, R.; Noll, B.; Pinkerton, M.H.; Taylor, R.; Madden, B.; Parsons, D.M. (2024). Distribution and potential causes of milky fleshed snapper in SNA 1. *New Zealand Fisheries Assessment Report 2024/25.* 57 p.

**TABLE OF CONTENTS**


**EXECUTIVE SUMMARY** **1**

**1. INTRODUCTION** **3**

1.1 Snapper distribution and diet 3

1.2 Milky flesh and poor body condition in fishes 3

1.3 Objectives 5

**2. METHODS** **5**

2.1 Condition classification 5

2.2 Interviews 5

2.3 Incidence of milky flesh snapper 6

Spatial and temporal analysis 6

Length analysis 7

2.4 Environmental correlates 9

2.5 Toxic substances 9

**3. RESULTS** **9**

3.1 Questionnaire responses 9

3.2 Quantitative analysis 11

Spatial and temporal analysis 11

Length analysis 17

3.3 Environmental correlates 17

Southern Oscillation Index 17

Trenberth 18

Satellite data: Chl- a, SST, and TSS 19

Freshwater input 22

Biological recruitment 25

Parasites 26

3.4 Toxins 27

**4. CONCLUSIONS** **28**

4.1 Impacts 29

**5. POTENTIAL RESEARCH** **30**

**6. ACKNOWLEDGEMENTS** **30**

**7. REFERENCES** **31**

**APPENDIX 1: Satellite data** **36**

**APPENDIX 2: Review of potential toxicity effects** **40**

Background information 40

Toxic metals 41

Legacy persistent organic pollutants 43

Current-use pesticides 46
Emerging organic contaminants 48
Causality assessment 49
Recommendations 54

**APPENDIX 3** **57**

**Plain language summary**

Commercial and recreational fishers were seeing snapper in the SNA 1 Quota Management Area with white, mushy flesh rather than their usual translucent, firm flesh. These fish were identified as being malnourished and this syndrome was called the Milky White Flesh syndrome.

To work out how widespread the syndrome was, NIWA collected information from commercial and recreational fishers on whether a snapper had the syndrome and where it had been caught.

Environmental information, such as sea surface temperature, was also examined to see if any changes had occurred in the period before fish with the syndrome were seen.

Snapper with the syndrome were found in large numbers from east Northland down to the Hauraki Gulf. The Bay of Plenty was generally unaffected.

The combination of a long La Niña period with warm sea surface temperatures, low productivity (and therefore less food at the bottom of the food chain), and a big increase in the number of young snapper appears to have contributed to the syndrome.

Understanding where affected fish are and possible causes can help with effective monitoring of this important snapper stock.

# EXECUTIVE SUMMARY

**Johnson, K.S.[1]; Gadd, J.; Bian, R.; Noll, B.; Pinkerton, M.H.; Taylor, R.; Madden, B.; Parsons, D.M. (2024). Distribution and potential causes of milky fleshed snapper in SNA 1.**

***New Zealand Fisheries Assessment Report 2024/25.* 57 p.**

An investigation into the distribution of milky fleshed snapper (*Chrysophrys auratus*, SNA) in the SNA 1 Quota Management Area (QMA) was conducted in 2022 and 2023. Commercial samples from SNA 1 catch-at-age sampling, commercial grading data, recreational angler records, and filleting shop data were combined with an examination of environmental variables and a review of the potential for toxins to contribute to the syndrome. Information was analysed for the three SNA 1 population sub-regions: East Northland, Hauraki Gulf, and the Bay of Plenty.

Recreational and commercial data indicated that the regions with the highest incidence of milky fleshed snapper were the central inner Hauraki Gulf and East Northland, in particular around Doubtless Bay. The abundance of milky fleshed snapper increased during winter, particularly between June and August. Although an underestimate, 48.9 tonnes of skinny 'SKY' grade snapper were caught by commercial fishers between January and November 2023, indicating that these snapper constitute a substantial quantity of the commercial catch.

Fish with Milky White Flesh Syndrome were known to be in a state of chronic malnutrition. Testing by the Ministry for Primary Industries (MPI) Animal Health Laboratory showed that fish had poor body condition, and, at the cellular level, there was muscle degeneration, liver atrophy, and iron accumulation in the liver due to their severely malnourished state and tissues breaking down. However, the root cause of the syndrome remained uncertain.

There has been an extended La Niña event, causing warm conditions and reduced upwelling off the east coast of the North Island. The Hauraki Gulf has also had a sustained low chlorophyll-a concentration for the last decade. However, the Firth of Thames, East Northland, and the west coast of the upper North Island have had increasing chlorophyll-a concentrations over the last two years. The increased snapper biomass recruiting in the Hauraki Gulf combined with warmer conditions and low primary productivity may have contributed to the incidence of milky fleshed snapper in this region. However, it does not fully explain the incidence in East Northland, where recruitment has remained steady and chlorophyll-a concentrations have increased in recent years.

A review of various potential toxins revealed a paucity of monitoring in sites around the SNA 1 QMA. The large spatial spread of milky fleshed snapper and the longevity of the effects make it unlikely that toxins are a causative agent; the quantity of a toxin needed to cause such an effect in a dynamic coastal environment is very large and unrealistic for any current use agricultural chemicals.

Parasites are also unlikely to be the cause of the milky fleshed snapper syndrome, as determined by MPI testing. Specifically, internal helminth parasites were present in affected fish, although similar parasite levels were also observed in 'healthy' snapper. Furthermore, the presence of the syndrome in species other than snapper suggests that parasites are an unlikely cause due to host-specificity of many parasites. MPI testing also noted no evidence of a septic or infectious (viral, bacterial, fungal, or protozoal) cause for the muscle degeneration.

It is likely that a combination of factors have caused the milky fleshed snapper syndrome in SNA 1. Long-term low chlorophyll-a concentrations, less upwelling along the east coast, increased snapper biomass in the Hauraki Gulf, and an increased metabolic rate because of increased temperatures are

[1] All authors: National Institute of Water and Atmospheric Research (NIWA), New Zealand.

all likely to put strain on the snapper populations. However, when compared to the North Island west coast where milky fleshed snapper have not been observed, the environmental correlates examined do not fully explain the spatial distribution of milky fleshed snapper.

# 1. INTRODUCTION

In 2019, commercial fishermen in the inner Hauraki Gulf started reporting catching ‘skinny’ snapper (*Chrysophrys auratus*, SNA). Over the subsequent three years the prevalence of skinny snapper in commercial Hauraki Gulf catches was observed to increase markedly (Walsh et al. 2022). By August 2022, the Ministry for Primary Industries (MPI) and NIWA had also received multiple reports from recreational fishers of milky white fleshed (MWF) snapper being caught in the Hauraki Gulf, and the condition was named Milky White Flesh Syndrome.

Samples of affected fish were collected and tested by MPI’s Animal Health Laboratory. Results from testing showed that fish were in a state of chronic malnutrition; fish had poor body condition and, at the cellular level, there was muscle degeneration, liver atrophy, and iron accumulation in the liver due to their severely malnourished state and the breakdown of tissues. Although internal helminth parasites were present in affected fish, similar parasite levels were also observed in ‘healthy’ snapper. Consequently parasites, per se, are not thought to be a direct cause of the syndrome, although it was noted that the inflammation they cause could affect digestion, which would be more problematic for chronically malnourished fish. There was no evidence of a septic or infectious (viral, bacterial, fungal, or protozoal) cause for the muscle degeneration. The root cause of the MWF syndrome remained uncertain, prompting Fisheries New Zealand to commission this report which seeks to document the distribution of MWF snapper within the SNA 1 Quota Management Area (QMA), along the east coast of the North Island from North Cape to Cape Runaway.

The SNA 1 QMA constitutes the largest snapper fishery in New Zealand, both in terms of commercial and recreational catch, with recreational and commercial fishing activity occurring along the full length of the coast. As such, the prevalence and locations of MWF snapper are of interest as this syndrome has the potential to impact an important fishery; for example, commercial fishers have reported redistributing their fishing effort to avoid encountering large numbers of MWF snapper.

## 1.1 Snapper distribution and diet

There are three populations of snapper within SNA 1: East Northland, Hauraki Gulf, and the Bay of Plenty; with limited mixing between populations (Fisheries New Zealand 2023). Snapper can also have different residency modes: one a spawning population that is more mobile, and one a resident subpopulation of shallow reef dwelling individuals (Parsons et al. 2016). Snapper move to spawning grounds around November–December where they serially spawn, releasing several batches of eggs during spring and summer, before dispersing to inshore feeding grounds. After a short planktonic phase, larval snapper settle in nursery grounds in estuaries and harbours. Some adults move to deeper waters during winter, where they are more dispersed.

Snapper are opportunistic feeders, with their diverse diet largely consisting of crustaceans, polychaetes, echinoderms, molluscs, and other fish (Godfriaux 1969, 1974). Larval and post-settlement juveniles feed on zooplankton, primarily copepods and cladocerans, mysids, and crustaceans (Parsons et al. 2014), whereas larger snapper consume more hard bodied and larger prey items, such as fish, echinoderms, hermit crabs, molluscs, and brachyuran crabs (Godfriaux 1969, Usmar 2012).

## 1.2 Milky flesh and poor body condition in fishes

SNA 1 MWF snapper are reported to have milky, pale flesh with a ‘mushy’ texture, as well as being ‘skinny’ in appearance. The MPI case definition for ‘Milky white flesh syndrome (MWFS) in snapper (*Chrysophrys auratus*)’ refers to fish being skinny and in poor condition (A. Pande, pers. comm.). However, there are also reports of snapper that externally present as being healthy but are found to have milky flesh when internally examined. These may be affected individuals which are either deteriorating or recovering from a poor body condition.

It is important to note that not all fish with white flesh have MWF syndrome. White flesh can occur naturally after spawning in spring and early summer; although this white flesh can be soft, it does not have a mushy texture. Snapper show a cyclical increase in energetic reserves in their gonads and liver prior to their spawning period, with a marked decrease in gonadosomatic and hepatosomatic indices post-spawning (Crossland 1977, Scott & Pankhurst 1992). This is consistent with a decline in body condition immediately post-spawning when energy reserves have been depleted and are yet to be replenished. Fish recover condition post-spawning, and under normal conditions their flesh is firm and translucent again by late February. It should be noted that MWF syndrome can affect fish of all sizes and occurs throughout the year, rather than only affecting adult fish following spawning.

Prior to 2019, whilst fishers reported regularly seeing a decline in body condition and slightly white flesh in snapper around summer spawning, the incidence of MWF snapper (with mushy textured flesh) was a rare occurrence and was virtually never observed during winter months. MPI note reports of MWF Hauraki Gulf snapper in poor condition in 2012, although these reports were at a lower frequency than the current situation (M. Griffiths, Fisheries New Zealand, pers. comm.). Occurrence of MWF snapper were not subsequently reported again until 2019, suggesting that the fish affected in 2012 were no longer apparent in the population, either having recovered or died.

Poor body condition and changes in flesh quality and structure have been documented for a range of species. The appearance of a similar body shape and muscular structure to the histological laboratory testing results from affected snapper have been described from toothfish *Dissostichus mawsoni*, referred to as “axe handle” toothfish (Fenaughty et al. 2008), sockeye salmon *Oncorhynchus nerka* (Kiessling et al. 2004), and Atlantic cod *Gadus morhua* (Lambert & Dutil 2000). These changes were attributed to spawning; there was observed shrinkage in muscle fibres, with energy from lipids and protein going to both swimming effort for migration and lipid reserves being prioritised for gonad development (Kiessling et al. 2004, Lambert & Dutil 2000). The pathological analysis of affected snapper noted muscle atrophy and depleted or poorly developed gonads in several individuals (from an admittedly small sample). These individuals were submitted for analysis in November, a time of year when gonads would be expected to be reasonably well developed (Scott & Pankhurst 1992). Therefore, cyclical spawning changes do not appear to explain the poor body condition in snapper from SNA 1.

A similar phenomenon to MWF snapper has also been reported in Alaskan and Greenland halibut *Reinhardtius hippoglossoides*, referred to as Mushy Halibut Syndrome (MHS) (Meyers et al. 2019). The cause of this change in condition has been attributed to nutritional depletion (Ortega et al. 2023) although the specific factors underlying this were not reported. It was suggested that a diet shift following a decline in forage fish could have resulted in essential nutrients being missed (Zador & Yasumiishi 2017). Experimental selenium deficiency caused decreased weight gain and a shrinkage of muscle fibres in rainbow trout *Oncorhynchus mykiss* (Zhang et al. 2022) and ‘thin back disease’ in carp *Cyprinus carpio* (Wang et al. 2013).

A case similar to MWF snapper was observed in a population of black drum *Pogonias cromis* from a super saline lagoon in Texas; individuals had gelatinous fillets, reduced white muscle in the dorsal region, and some atrophy of organs (Olsen 2016). There was a particularly large emaciation event in 2012 during a drought year (Olsen 2014), and, although the cause of emaciation was not determined, it was thought that the hypersalinity due to the drought event resulted in resource restrictions to limit the diet of this species, including limiting the movement of this species out of their lagoon (Breaux et al. 2019). Monitoring of these fish suggested that the emaciated fish returned to a ‘normal’ body condition within 18 months (Olsen 2016), suggesting that this was a short-term occurrence likely coinciding with the change in environmental conditions. Poor body condition as a result of changes to environmental conditions (leading to malnutrition) has also been documented in *G. morhua* from the Baltic Sea (Eero et al. 2015, Casini et al. 2016, Limburg & Casini 2019) and striped seabass *Morone saxatilis* in upper Chesapeake Bay (Uphoff 2003).

## 1.3 Objectives

The overall objective of Fisheries New Zealand project SEA2022-08 was to document the spatial and temporal distribution of snapper with ‘milky flesh’ in the SNA 1 QMA. In addition, potential environmental drivers of the MWF syndrome in snapper were also investigated. This included a review of the potential for toxic effects in snapper.

# 2. METHODS

## 2.1 Condition classification

Cases of MWF snapper referred to in this report are predominantly suspect cases, as per the MPI case definition (A. Pande, pers. comm.), due to condition factors not being routinely calculated. Confirmed cases are only possible with laboratory bacteriology and histopathology testing to confirm that no other infectious process is present, and that muscle, liver, and other tissues show the expected degenerative signs (A. Pande, pers. comm). One of the criteria for a suspect case of MWF syndrome is having white flesh (Figure 1A), and consequently skinny snapper (those in poor body condition but that did not have their flesh examined; Figure 1C) did not meet these criteria. However, using only external assessments of snapper would likely have resulted in underestimates of the number of affected snapper because fish with externally good condition also presented with milky flesh.

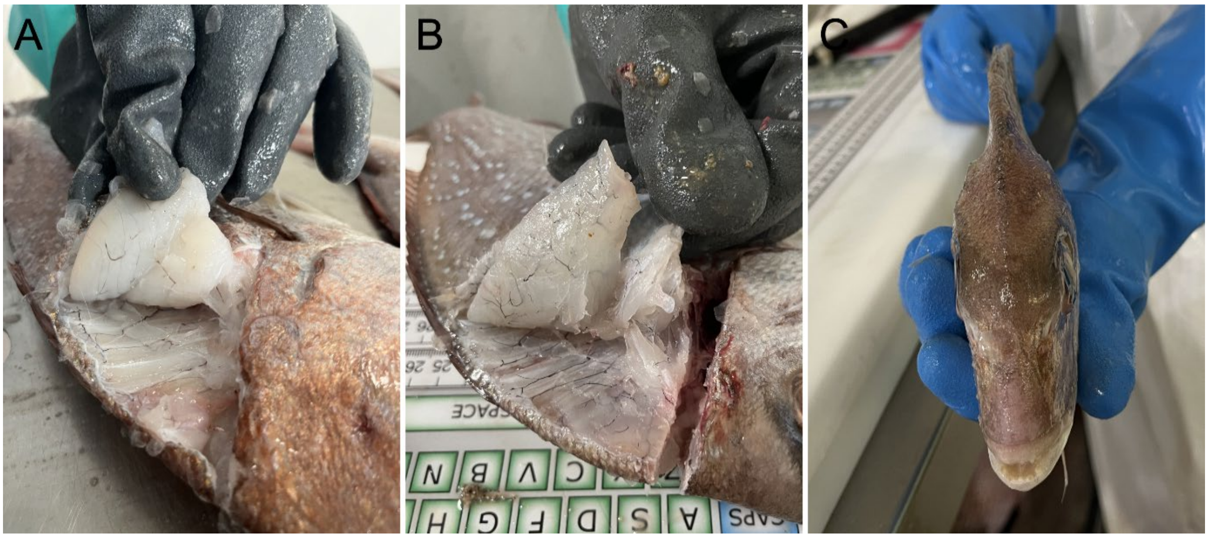



**Figure 1: Examples of (A) milky white flesh, (B) normal flesh, and (C) a skinny snapper with poor body condition. Photos: Rikki Taylor (NIWA).**

## 2.2 Interviews

Commercial and recreational charter skippers, shore-based Licensed Fish Receiver (LFR) staff, and recreational anglers were interviewed to determine the spatial occurrence of MWF snapper, how the occurrence may be changing over time, and to determine changes in fishing behaviour in relation to encountering these fish. A total of nine responses were collected, with some of these responses being a collective answer from numerous people, e.g., from all LFR staff. Most interviews were conducted by emailing companies a list of open-ended questions (Table 1), with some interviews conducted over the phone. Phone interviews were structured around the questions in Table 1, with the option to glean further information on any points of interest raised. Recreational charter skippers and public anglers were from the Hauraki Gulf, so responses are biased towards this region of SNA 1.

All interview responses were collated, analysed, and allocated themes to determine any commonalities among responses. Additional information that did not specifically relate to questions asked was also recorded for separate analysis.

**Table 1: Questions asked of commercial and recreational skippers, fisheries staff, and fishers.**

| Theme | Question |
|---|---|
| Start date | When did you start noticing the skinny/white fleshed fish turning up in the catch? |
| Location | Where were these fish being caught? |
| Quantity | What percentage of the catch did they make up? |
| Size | Was any particular size range of snapper affected? |
| Temporal changes | How has all of the above changed since you started noticing it (i.e., did the affected fish become more or less common, has the location they are being caught changed, has the size of affected fish changed)? |
| Vessel movements | Have vessel movements changed as a result of the capture of affected fish? |
| Previous instances | Is this something you have noticed in the past, but maybe at a lower frequency? |
| Causes | Do you have any thoughts as to what is causing the increased frequency of skinny/milky fleshed snapper? |

## 2.3 Incidence of milky flesh snapper

### Spatial and temporal analysis

Information on proportions of MWF snapper were collected from recreational anglers and local bait shops providing a filleting service. The Kai Ika filleting station at Westhaven marina (Figure 2) collected data on fishing location from customers, and noted incidence of MWF snapper during filleting from January to April and November to December 2023. Recreational anglers from the Mairangi Bay Fishing Club recorded MWF snapper incidence, fishing location, and substrate data from January to December 2023. These data were combined to provide a larger dataset for spatio-temporal analysis.

A public survey was created by LegaSea to collect information about MWF/mushy fleshed snapper and other species. The survey was open to the public during September and October 2023, and respondents were asked to record details from their last fishing trip only. Information on the presence and location of MWF snapper was used to examine numbers of respondents encountering MWF snapper in SNA 1. Other species and locations were noted.

Two commercial fishing companies that operate in SNA 1, Lee Fish and Moana, provided weight and landing data for total snapper caught per event and the weight of snapper that had been given a skinny grade ‘SKY’. It is worth noting that these fish were not assessed for flesh colour as they were not cut open. Commercial SKY grading provided data covering the period 16 January to 27 November 2023. Extracted data were initially groomed and checked for obvious errors. The vessel name, landing date, and greenweight of snapper were used to match SKY data to data extracted from the Fisheries New Zealand Enterprise Data Warehouse to determine the location within SNA 1 that SKY snapper were caught. When matching the data, vessel names needed to match exactly, the difference between landing dates needed to be within two days, and any difference between greenweights needed to be smaller than 2%. This process matched 91% of trips automatically. Further manual matching of trips was not conducted due to time constraints.

Although trips that caught SKY snapper were identified, events within each trip were unable to be accurately attributed to the SKY catch. The process followed was to prorate the SKY weight to events within a trip based on their estimated catch weights. Any trips that crossed the SNA 1 boundary were excluded from the dataset. Trips that covered a large area within SNA 1 were examined for influence on results and to ensure that SKY location data remained accurate. Removing these trips made negligible difference to the SKY locations and weights, so all trips that covered a large area were retained in the dataset. The weight and percentage of SKY grade catch were plotted by meteorological season: January–February; March–May; June–August; September–November.

It was originally intended that NIWA shed sampling of ungraded landings from SNA 1 (Fisheries New Zealand project SNA2022-01) would also be used to document the proportion of MWF snapper. These data, however, were not used. A total of 262 landings were sampled as part of the SNA2022-01 project, but many of these samples took place before the current milky fleshed snapper project was initiated and a protocol to assess skinny/not skinny status was developed. In addition, for bottom longline (BLL) landings, this status was only assessed for a maximum of 35 fish per landing. Furthermore, it is known that fishers were actively avoiding the areas with highest proportions of skinny snapper. As a result, not all landings sampled contained skinny fish and, for the landings that did, skinny fish comprised a small proportion of the catch. Overall, shed sampling offered ineffective sample sizes to assess the proportion of skinny fish and subsequent analysis was deemed uninformative (although this sampling was used to conduct analysis of the length composition of skinny vs. non-skinny snapper, see below). Conversely, the commercial SKY grading process covered every landing and every fish (since January 2023 at Moana and June 2023 at Lee Fish), resulting in much larger sample sizes with less likelihood of missing skinny fish, other than those missed by fisher movement away from the worst affected areas.

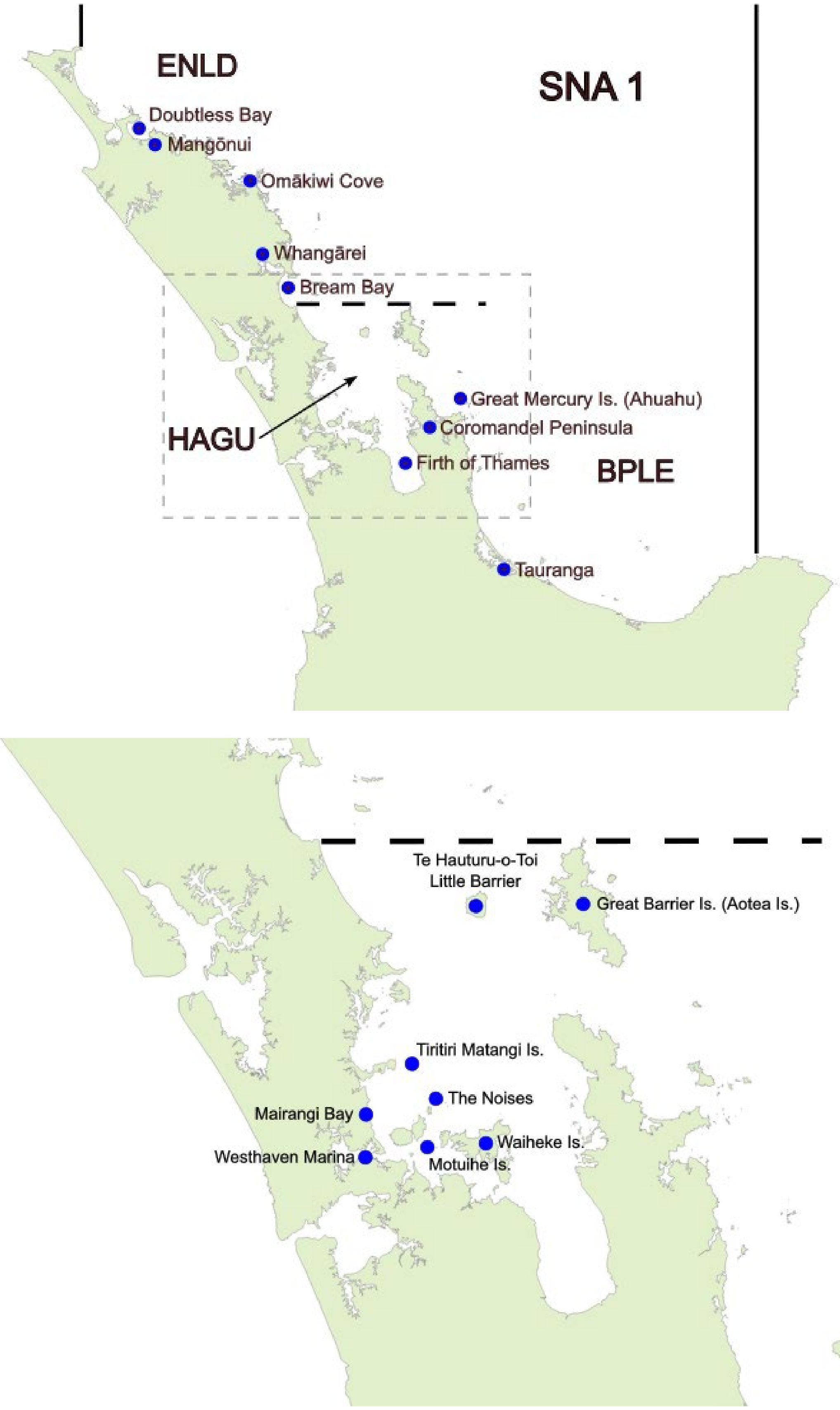
A bait shop on the Coromandel Peninsula, Salty Towers, collected incidence data on MWF snapper from the filleting service they provided. This data set covered the period between March 2021 to December 2023 inclusive. Initial records of MWF snapper noted ‘white fish’ within a large batch of fish rather than specific numbers. Due to the lack of detail, these instances were recorded here as a single MWF snapper and consequently the proportions of MWF snapper presented are underestimates, particularly for earlier months in the time series. These data were likely from fish predominantly collected in the Firth of Thames (specifically in association with mussel farms), although fishing location data were not collected. Incidence of MWF snapper was analysed as a time series.

It should be noted that all these sources would provide underestimates of MWF snapper occurrence. Recreational anglers returned fish that were very skinny and moved from the worst affected areas. Fish filleting services would only be given fish deemed sufficient quality to fillet. Therefore, the very skinny snapper, and potentially the worst affected, would be excluded from these data. In addition, commercial fishers have been actively avoiding areas with MWF snapper and grading data from commercial fishing companies are also underestimates; fish that are deemed too skinny to fillet are rejected, with only those deemed suitable for sale graded as ‘SKY’. Therefore, fish in very poor condition were excluded from weights provided. The commercial data also excluded affected fish that did not appear skinny.

### Length analysis

During the NIWA shed sampling described above, length data for 5252 ungraded snapper were recorded, which were also externally assessed to determine whether they were classed as ‘skinny’ (i.e., fish were assessed prior to the commercial grading process which determines SKY grade catch). Key features used for this determination were: sunken flesh below brow ridge and thin brow ridge when viewed from above; apparent ‘bulging’ eye due to sunken flesh above; lack of girth before the tail. Any fish that were borderline were not classed as ‘skinny’ to avoid misidentification. Due to being an external assessment, these fish could not be confirmed as having milky flesh.

(a)




(b)

**Figure 2: Map of SNA 1 Quota Management Area with key locations identified. Grey dashed box in map (a) indicates Hauraki Gulf area enlarged in map (b). ENLD: East Northland, HAGU: Hauraki Gulf, BPLE: Bay of Plenty.**

## 2.4 Environmental correlates

The following environmental correlates likely to influence fish condition were selected: Southern Oscillation Index (SOI); Trenberth Z1 index; chlorophyll-a concentrations (Chl-a); sea surface temperature (SST); total suspended solids (TSS); and freshwater input, specifically rainfall and river flow.

The SOI quantifies the pressure difference between Darwin, Australia and Tahiti, providing an indicator of El Niño-like and La Niña-like conditions. The Z1 Trenberth circulation index is the pressure difference between Auckland and Christchurch relative to 'normal' and provides an indicator of west-east airflow (Trenberth 1976). The SOI, Trenberth, and rainfall data were each collated from the NIWA climate database CLIDB and averages plotted for January 2010 to November 2023.

Satellite data were used for the same time period for Chl-a, SST, and TSS. Chl-a was determined from locally-received downlink data (SCENZ v4.0 CHL) and products from the Copernicus project which uses multiple sensors; SST used the Optimum Interpolation Sea Surface Temperature (OISST) product version 2.1; and TSS was calculated using SCENZ TSS v3.0 product and products from Copernicus (for further details of satellite data processing and products, see NIWA-SCENZ website). Monthly anomalies were calculated as the average value for each month, minus the long-term monthly mean (i.e., the mean for that particular month across all years). The reference periods against which the anomalies were judged differed for the three environmental variables; for SST, the reference period matched the SeaWiFS-MODIS time series (complete years 1998–2022) which used the OISSTv2.1 data, and the reference periods for Chl-a and TSS matched the period of operation of the MODIS-Aqua sensor (complete years 2003–2022).

The Piako River and Waihou River were selected to examine freshwater input as they are the two largest rivers feeding into the Hauraki Gulf. River flow and level data for the Piako and Waihou rivers was supplied by Waikato Regional Council. Measurements between 1 January 2010 and 11 December 2023 were used: the Piako River measurements were taken from the Paeroa-Tahuna Road Bridge station; and the Waihou River measurements were taken from the Te Aroha station.

Biological recruitment data for snapper between 1970 and 2022 in East Northland, Hauraki Gulf, and Bay of Plenty were obtained from the Fisheries New Zealand plenary report for SNA 1 (Fisheries New Zealand 2023). This recruitment index was used as a proxy for determining changes in snapper biomass throughout SNA 1.

In late 2023 there were reports of abrasions and parasites on snapper from recreational and commercial fishers. These parasites were identified and are discussed with respect to their impact on MWF snapper.

## 2.5 Toxic substances

The literature of existing knowledge of toxic substances was reviewed (Appendix 2). Specifically, toxic metals, legacy persistent organic pollutants, current-use pesticides, and emerging organic contaminants were analysed for known toxicology and occurrence. A causality assessment was carried out to determine the strength of evidence for the candidate causes, with recommendations for next steps based on the outcome of this assessment.

# 3. RESULTS

## 3.1 Questionnaire responses

Although information was gathered from a range of sources, there were some common themes among responses (Table 2). The majority of respondents first identified MWF snapper in winter 2022, with some respondents noting incidences from winter 2021. One respondent also noted that they have become more prevalent over the past 10 years, suggesting that they have been present for some time in lower numbers than are observed currently.

Responses regarding location and the size range affected were generally very consistent, with the worst affected area noted as the Inner Hauraki Gulf in shallower water, with mainly small (300–400 mm fork length, FL) fish being affected. Beyond approximately 30 m depth, the prevalence of MWF snapper was noted to decrease. These parameters match the known movements of young snapper, moving towards the Inner Hauraki Gulf to shallower waters. One respondent interviewed in November 2023 noted that in late October and early November 2023 MWF snapper were observed towards the outer Hauraki Gulf, on the eastern side of Great Barrier Island (Aotea Island), where they had not been encountered previously. Furthermore, there was an increase in the number of larger individuals affected, with 'razor-back' individuals up to 550 mm FL becoming more common. This suggests that there are potential changes in the distribution of affected snapper over time.

It is difficult to determine any changes over time from interview responses as most respondents said that they moved fishing locations to avoid areas with higher incidence of MWF snapper. This movement of fishing location also contributed to the wide range in the percentage of catch that were milky fleshed. Although all respondents noted the presence of thin fish previously, generally attributed to spawning, several respondents noted that the current MWF snapper appeared different from those encountered in the past; those previously encountered did not have such white flesh, and mushy flesh had never been observed. This suggests that the degradation of muscle in the current MWF snapper population is somewhat different to spawning-induced loss in condition and previous occurrences. Furthermore, the fact that there are seemingly healthy fish (with externally good body condition) also being observed with white and sometimes mushy flesh supports the suggestion that the current pathological changes are different from those observed in the past.

**Table 2: Themes identified in questionnaire responses from commercial and recreational skippers, shore-based workers, and anglers regarding milky fleshed snapper.**

| Question | Most common response |
|---|---|
| Date of first incidence | Winter 2022 |
| Location caught | Inner Hauraki Gulf, in shallower water |
| Percentage of catch | Around 10% but variable (<1% to 100%) |
| Size range affected | Small, 300–400 mm FL |
| Any changes since first seen | Minimal change seen (but less fishing in areas where prevalent) |
| Changes in vessel movements | Moved away from affected area/s, often to deeper water |
| Past incidences | Yes<br>- Attributed to spawning<br>- Fish were skinny and milky, but not mushy |
| Thoughts on causes<br>(*listed in order of frequency of response*) | Lack of food<br>Run-off / Sedimentation<br>Increased fish biomass<br>Higher temperatures<br>La Niña<br>Spawning |

Anecdotal comments indicated that MWF snapper were leaving the Hauraki Gulf around June 2023, both heading to deeper water and moving further north; commercial skippers based in Mangōnui and Whangārei reported presence of 'skinny' snapper in catches in June 2023. Additionally, MWF snapper appear to be most abundant over sandy or muddy substrate, with lower incidence observed around rocky reefs. This observation is supported by the quantitative data presented in Section 3.2, where MWF snapper contributed 24.2% and 4.2% of catch over soft sediment and rocky reef habitat, respectively, from Mairangi Bay Fishing Club angling data between January and November 2023. Comments on body condition noted that affected snapper were typically lacking in colour compared with healthy fish, with some losing their scales more readily than their seemingly healthier counterparts.

## 3.2 Quantitative analysis

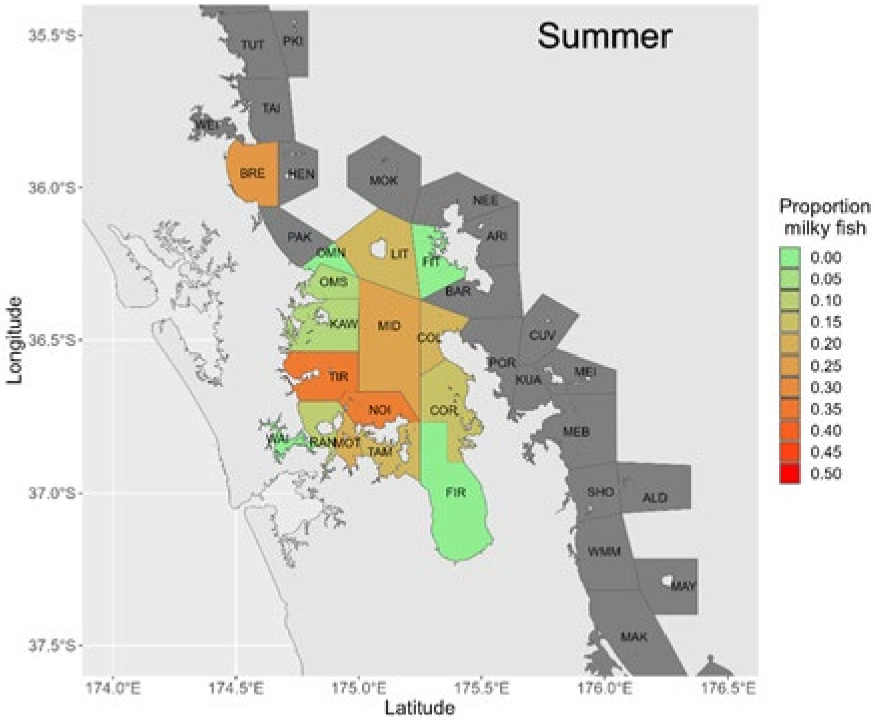
### Spatial and temporal analysis

The central Hauraki Gulf recorded the highest proportion of recreationally caught MWF snapper, particularly between Waiheke, Te Hauturu-o-Toi Little Barrier, and Tiritiri Matangi islands (region codes NOI, MID, LIT, TIR; Figure 3). There was a higher proportion of MWF snapper in the inner Hauraki Gulf to the south of Waiheke Island and around Motuihe Island (TAM and MOT) during summer months, although this could be a reflection of the larger sample size for this time period, i.e., more recreational fishers are fishing in the Hauraki Gulf during summer months (and fewer fishers during winter) meaning that small changes in milky fleshed fish encountered during winter had a large effect on average proportions per region. Tiritiri Matangi (TIR) and the Noises (NOI) areas north of Waiheke Island each recorded 33% of catch as milky during the full survey period, with the majority of these MWF snapper caught during summer. Whilst the worst affected areas appear to be slightly further north during winter, it is difficult to draw conclusions due to the large discrepancy in sample sizes between summer and winter (n = 126 and 39, respectively), as well as the overall small sample size.

Recreational anglers' responses from the LegaSea online survey showed that the area with the highest incidence of MWF snapper was in the central Hauraki Gulf north of Waiheke Island (region MID; Figure 4). This was consistent with the location identified by the proportional recreational angler data. The LegaSea survey showed a higher number of MWF snapper in the Firth of Thames (FIR) than the proportional data showed. As respondents were asked to record information about their latest fishing trip only, over 75% of records were from August and September 2023, so it was not possible to make any temporal comparisons. Despite mushy flesh not being used as a criterion, records from August and September would not have included fish that had white flesh as a result of spawning. Recreational anglers were asked to confirm how they identified the fish as having milky flesh: 95% of anglers identified snapper as milky by filleting, and the remaining anglers looked at a fish, noted how thin it was, and released it.

There were several reports of MWF snapper from East Northland, particularly around Bream Bay and Doubtless Bay. There were only a small number of records from the Bay of Plenty. Although not mapped due to lack of specific location information, respondents noted presence of MWF snapper along the west coast of the North Island in low numbers: in west Northland (1 respondent), Kaipara Harbour (5), Manukau Harbour (2), and in the Taranaki region (1). All fish except one from Kaipara Harbour were identified as MWF by filleting. Although not central to this research, there was a range of other species recorded as having milky flesh (Table 3).

(a)




(b)

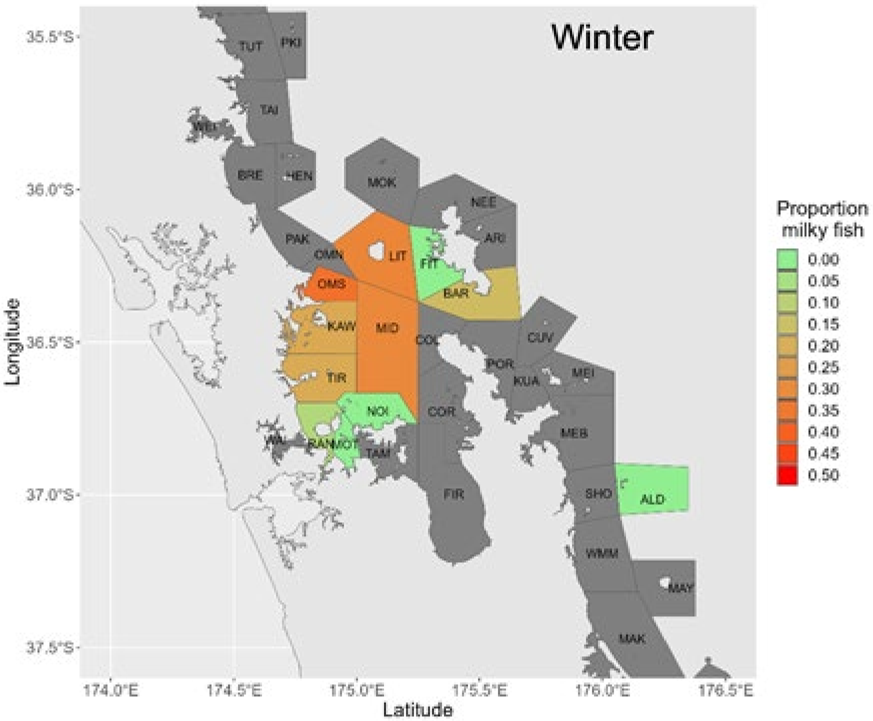



**Figure 3: Proportion of milky fleshed snapper recorded per event by recreational anglers from the Mairangi Bay Fishing Club and at the Kai Ika filleting station, Westhaven, between 1 January and 31 December 2023. (a) Summer: October to April (n = 126), (b) Winter: May to September (n = 39). Areas coloured grey have no data.**

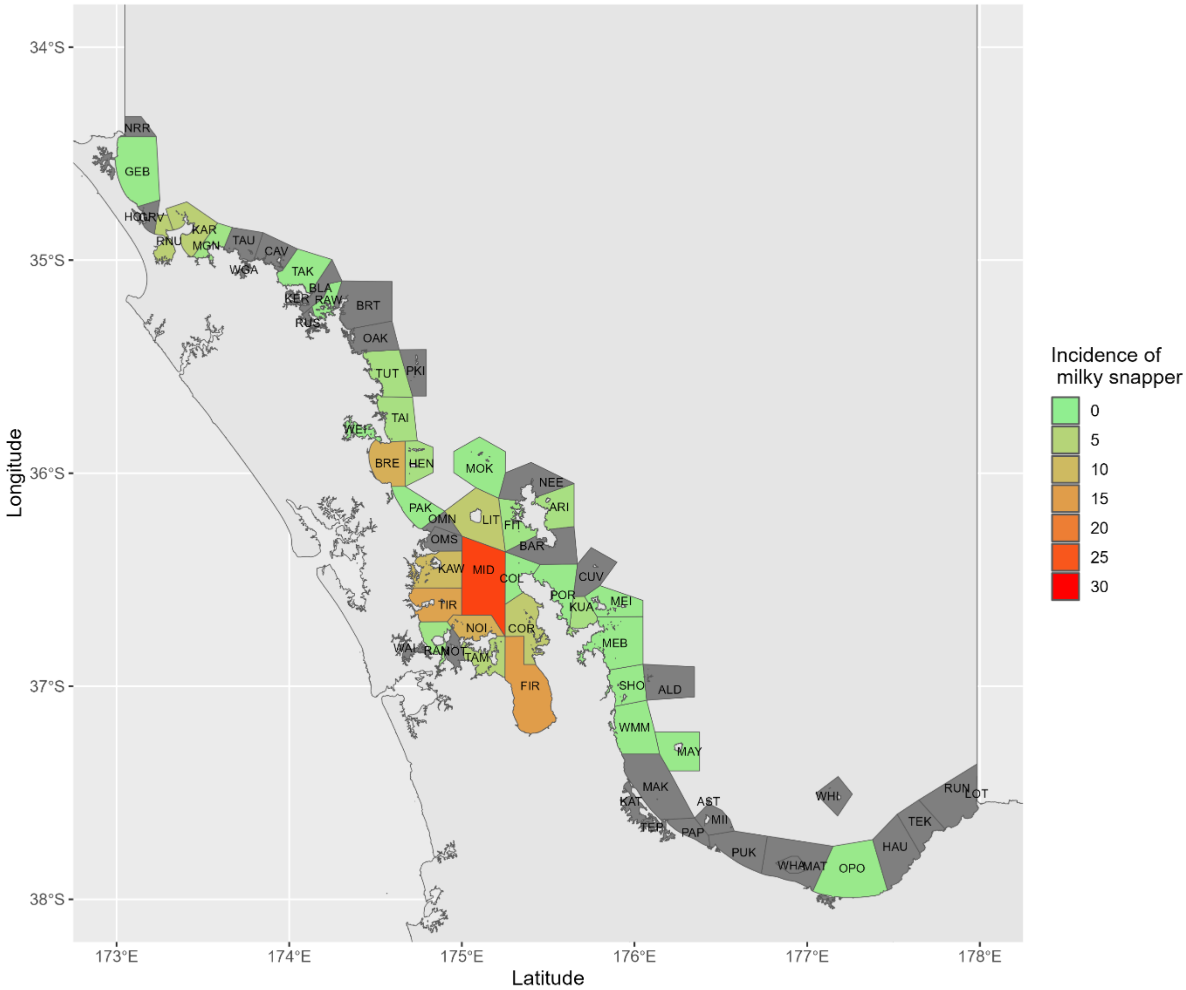



**Figure 4: Location of milky fleshed snapper recorded between December 2022 and October 2023, for recreational fishers' latest fishing trip only. Each incidence represents one respondent noting that they encountered milky fleshed snapper. Areas coloured grey have no data. Data supplied by LegaSea.**

**Table 3: Non-snapper species recorded as having milky flesh by recreational anglers. All fish were identified as having milky flesh by filleting, apart from one trevally which was identified by external examination and released. Data supplied by LegaSea.**

| Species | Number of respondents | Region caught |
|---|---|---|
| Trevally<br>*Pseudocaranx dentex* | 10 | Eastern Northland, Hauraki Gulf, Bay of Plenty |
| Kahawai<br>*Arripis trutta* | 10 | Hauraki Gulf, Bay of Plenty, Manukau Harbour, Taranaki |
| Red gurnard<br>*Chelidonichthys kumu* | 3 | Hauraki Gulf |
| Kingfish<br>*Seriola lalandi* | 1 | Bay of Plenty |
| Mahi mahi<br>*Corphaena hippurus* | 1 | West coast Auckland |

Skinny snapper were given their own commercial grade ‘SKY’ from January 2023 and June 2023 respectively by two fishing companies. The location of fishing trips where SKY snapper were recorded spans the full SNA 1 fishery, with a higher proportion in the Hauraki Gulf and East Northland, in particular around Doubtless Bay (General Statistical Area 002). The proportion of skinny snapper was highest in the June to August catches. There was little change in the proportions of skinny snapper caught around the Coromandel Peninsula (particularly General Statistical Area 008) between March to May and June to August periods, whereas there was a marked increase in the proportion caught in the inner/central Hauraki Gulf (General Statistical Area 006) and East Northland (General Statistical Area 002) between these two periods. This is consistent with anecdotal interview responses noting skinny snapper in East Northland in June 2023. The overall lower proportion of skinny snapper in the Bay of Plenty region (General Statistical Areas 009 and 010) is particularly evident when comparing the total snapper fishery catch (i.e., including catch with no SKY graded fish) to the SKY grade catch over the full January to November 2023 period (Figure 5).

A total of 48.9 tonnes of SKY snapper were reported caught by these two companies between January and November 2023 in SNA 1. While this only constitutes 1.3% of the total catch, it is known to be an underestimate for several reasons. Snapper that were very skinny and unable to be filleted were rejected rather than graded as ‘SKY’. One commercial operator only introduced the SKY grade in June 2023, excluding fish that would have been SKY from earlier in the year. In addition commercial fishing vessels changed their fishing locations slightly to avoid the worst areas (see Section 3.1) and our analysis matched only 91% of SKY records and further manual matches were not conducted. With the addition of skinny snapper from these sources, it suggests that there were many more skinny snapper (likely MWF) present in 2023 than the fishery SKY weights and percentages suggest.

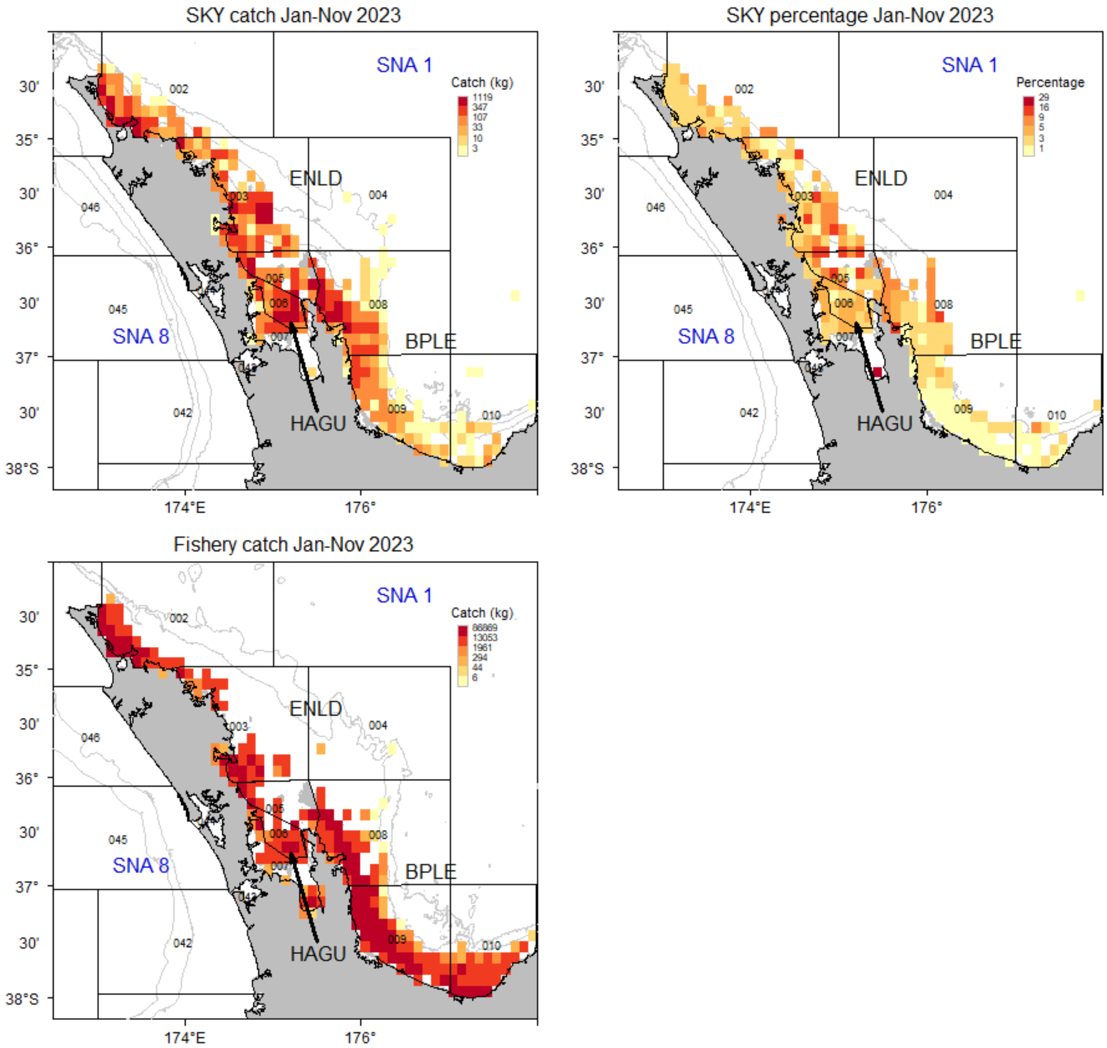



**Figure 5: Total catch weight of commercial snapper catch graded as ‘skinny’ in SNA 1 (top left) and percentage of the catch graded as ‘skinny’ (top right) from landings where SKY grade was recorded between January and November 2023, inclusive. The total fishery catch, including catch with no SKY graded fish, during the same time period is included for comparison (lower panel). Any pixel with fewer than three vessels or permit holders was removed from the total fishery plot. ENLD is East Northland, BPLE is Bay of Plenty, and HAGU is Hauraki Gulf.**

During 2021 and 2022, Salty Towers filleting service observed small numbers of MWF snapper from the Firth of Thames (FIR), with a small increase in proportion during July 2022 (2.6%) (Figure 6). There was a further small increase in the proportion of MWF snapper in March 2023, with the main peak in prevalence from June to September 2023, peaking in July 2023 at 17.4%. The large increase in MWF snapper noted in August 2022 from other sources was not reflected in this time series, suggesting that the incidence of MWF snapper may have been lower in the Firth of Thames than in the inner Hauraki Gulf during 2022.

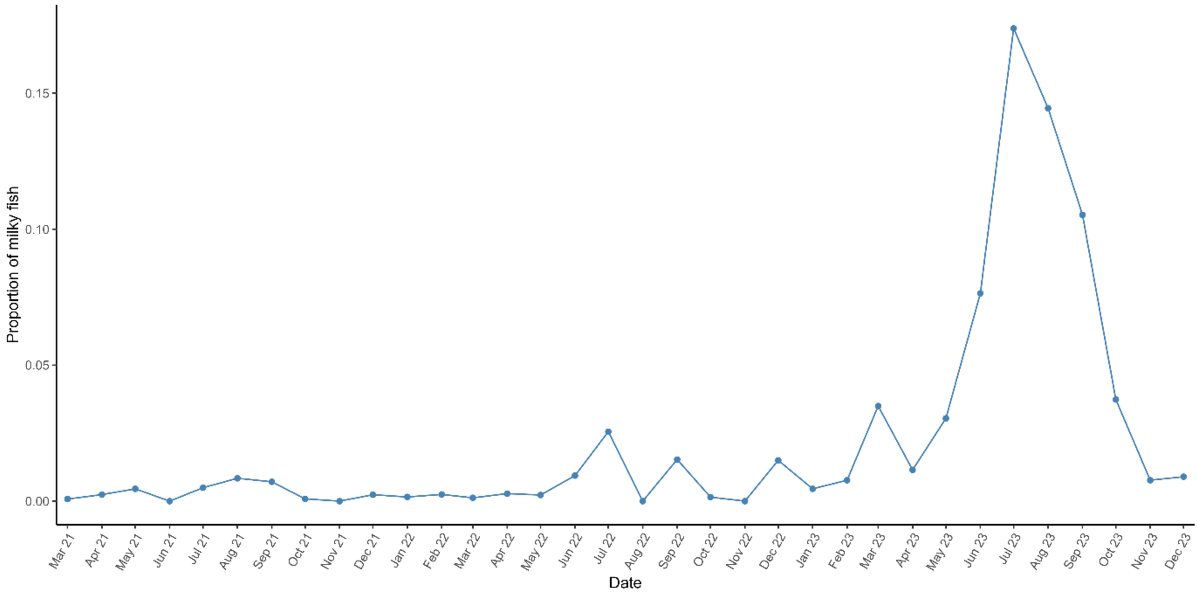



**Figure 6: Proportion of milky fleshed snapper observed when filleting Firth of Thames fish at Salty Towers between 1 March 2021 and 31 December 2023.**

The proportion of MWF snapper reported from the Kai Ika filleting station and recreational anglers from the Mairangi Bay Fishing Club in the Hauraki Gulf was variable between January and December 2023, ranging from 11.3% to 33.6% of catch each month (Figure 7). There appeared to be a steady increase in the proportion of MWF snapper between January and April, with further increases in June and September. Unlike the Salty Towers trend, this proportion decreased in July. The overall proportions observed from Kai Ika and Mairangi Bay Fishing Club were higher than those observed at Salty Towers, consistent with the results from spatial analysis showing higher proportions in the inner Hauraki Gulf than in the Firth of Thames.

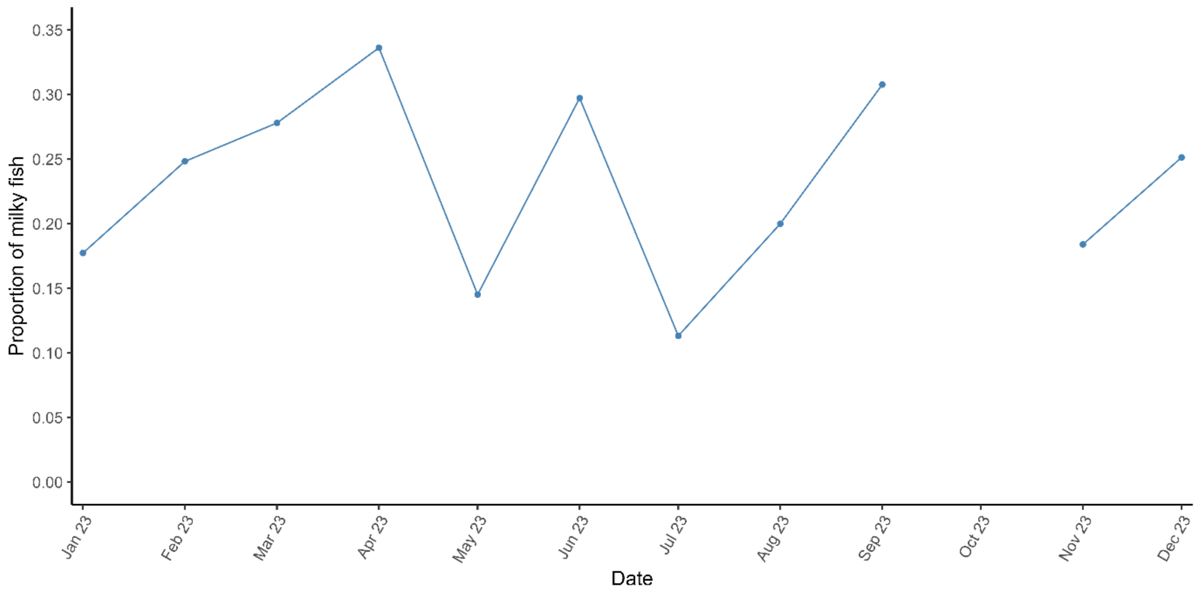



**Figure 7: Proportion of milky fleshed snapper recorded at the Kai Ika filleting station and by recreational anglers from the Mairangi Bay Fishing Club in the Hauraki Gulf between 1 January and 31 December 2023.**

### Length analysis

Skinny snapper were more constrained in their length range ($L_{min}$: 250 mm; $L_{max}$: 480 mm) than the regular snapper ($L_{min}$: 190 mm; $L_{max}$: 760 mm) (Figure 8). However, the general length distributions appear similar between skinny and regular snapper, with the large discrepancy in sample size (regular snapper: n = 5112; skinny snapper: n = 140) explaining the smoother curve of the regular snapper and the narrower range in the skinny snapper length frequency distribution. Median and mean lengths were similar between skinny (median: 320 mm; mean: 329.8 mm) and regular (median: 320 mm; mean: 327.7 mm) snapper, further supporting a similar size distribution between the two.

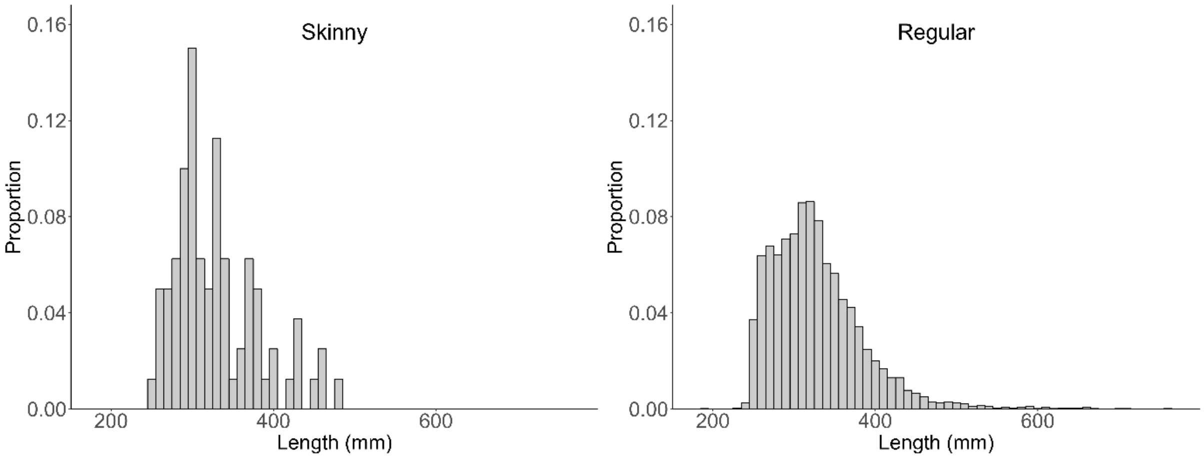



**Figure 8: Length frequency distribution of skinny and regular snapper in commercial SNA 1 catch, as classified by NIWA fisheries staff during snapper catch-at-age sampling (Skinny snapper: n = 140; Regular snapper: n = 5112).**

## 3.3 Environmental correlates

### Southern Oscillation Index

Over the past decade there have been oscillating positive and negative SOI events. During 2021 and 2022 there was a slightly prolonged La Niña-like event, with 32 consecutive months of positive SOI recorded between July 2020 and February 2023 inclusive (Figure 9). This longer period of positive SOI is associated with warmer air and water temperatures and stronger easterly winds around the North Island of New Zealand, resulting in less upwelling on the east coast (Zeldis 2004).

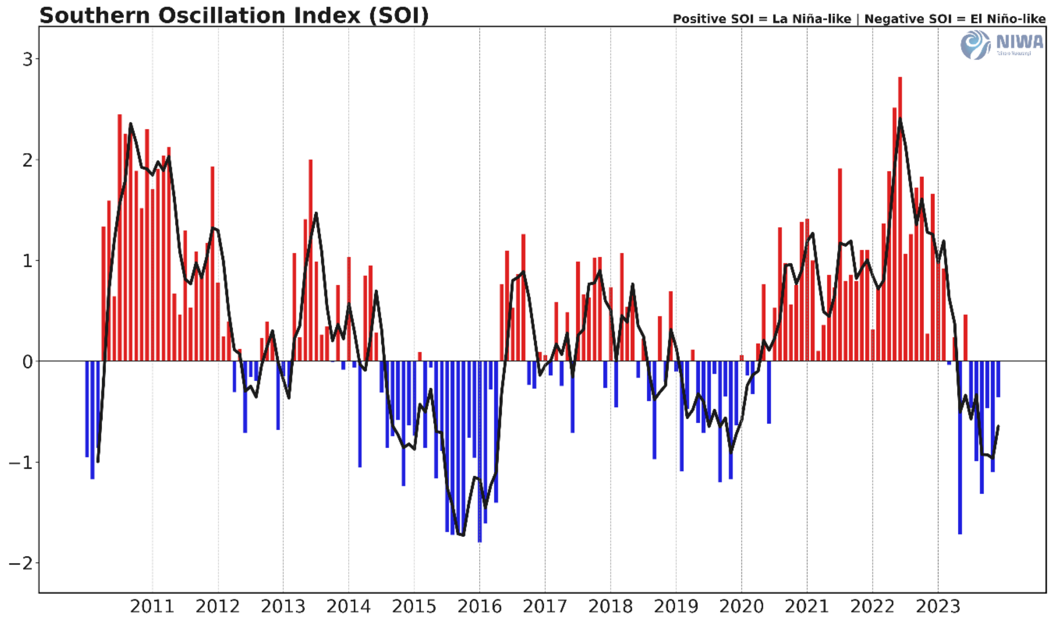



**Figure 9: Southern Oscillation Index between 1 January 2010 and 30 November 2023. Positive values indicate La Niña-like events and negative values indicate El Niño-like events. Each bar represents the monthly value and the black line shows the three-month rolling average. Grey vertical lines indicate January of the corresponding year.**

## Trenberth

The Z1 Trenberth index shows that there has been regularly oscillating positive and negative Z1 since 2010, indicating oscillating westerly winds and reduced westerly/easterly winds, respectively. October 2021 to April 2022 had a slightly larger negative Z1 than the preceding decade (Figure 10), indicating stronger easterly or reduced westerly winds during that period. January 2023 also recorded a much larger negative Z1 than the preceding decade. Reduced westerly winds during these periods result in reduced upwelling on the east coast of the North Island, suggesting periods with lower nutrient input.

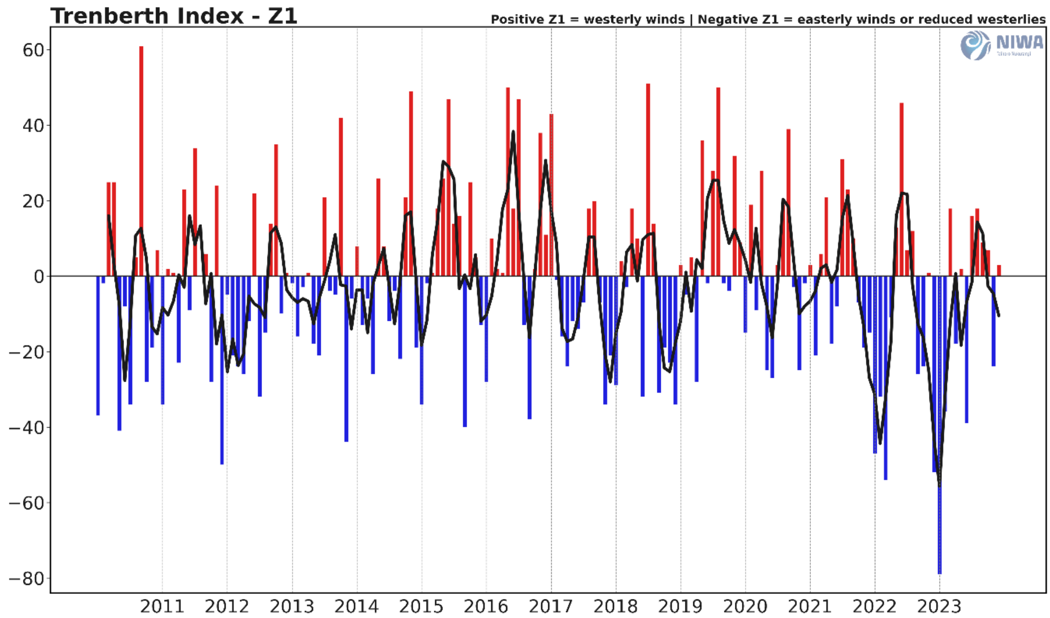



**Figure 10: Z1 Trenberth Index between 1 January 2010 and 30 November 2023. Positive values indicate westerly winds, and negative values indicate easterly or reduced westerly winds. Each bar represents the monthly values, and the black line shows the three-month rolling average Z1 index. Grey vertical lines indicate January of the corresponding year.**

### Satellite data: Chl- a, SST, and TSS

Chlorophyll-a concentrations have been sustained at low levels in the Hauraki Gulf. East Northland has seen a steady increase since 2019, whereas the Firth of Thames and the North Island west coast have had a more rapid increase from 2022 onwards (Figure 11, Appendix 1). These trends suggest that primary food sources may have been more limited in the inner and outer Hauraki Gulf than in the other regions over the last few years, potentially affecting zooplankton species composition and abundance.

Sea surface temperature has shown a steady increase over the last 20 years in all five regions examined: East Northland, Inner Hauraki Gulf, Outer Hauraki Gulf, Firth of Thames, and the upper west coast of the North Island (Figure 12, Appendix 1). There is little difference in trends between these regions. The peaks in the anomalies in 2022 are consistent with the 'unprecedented' marine heatwaves documented at the Leigh Marine Laboratory during 2022 (Shears et al. 2023). This increased temperature may not prevent snapper from feeding or reproducing, as snapper are not stressed by temperatures in the range experienced in the Hauraki Gulf (Bowering et al. 2023), but the increased metabolic rate as a result of increased temperature would increase their energy demands (Volkoff & Rønnestad 2020, Alfonso et al. 2021, Fangue et al. 2021). Morrongiello et al. (2021) demonstrated that snapper increased growth with increased temperature, further supporting that warmer conditions would lead to increased food demand. The process of converting food into energy is also affected by temperature, which suggests that increased temperatures lead to less efficient food conversion (Little et al. 2020). These factors together suggest that the increased SST observed may cause increased feeding rates in snapper, resulting in increased pressure on resources to sustain their increased energy demands.

There has been a generally decreasing trend in total suspended solids in all areas examined (Appendix 1), suggesting that water clarity has improved over the past decade. The larger decrease for the Firth of Thames and the North Island west coast between 2019 and 2023, observed as a steeper trend line in anomalies, may have contributed to the increased Chl-a concentrations with clearer water assisting with photosynthesis. However, it may be expected that with increasing Chl-a concentrations, the TSS would begin to decrease again, which is not evident in these data.

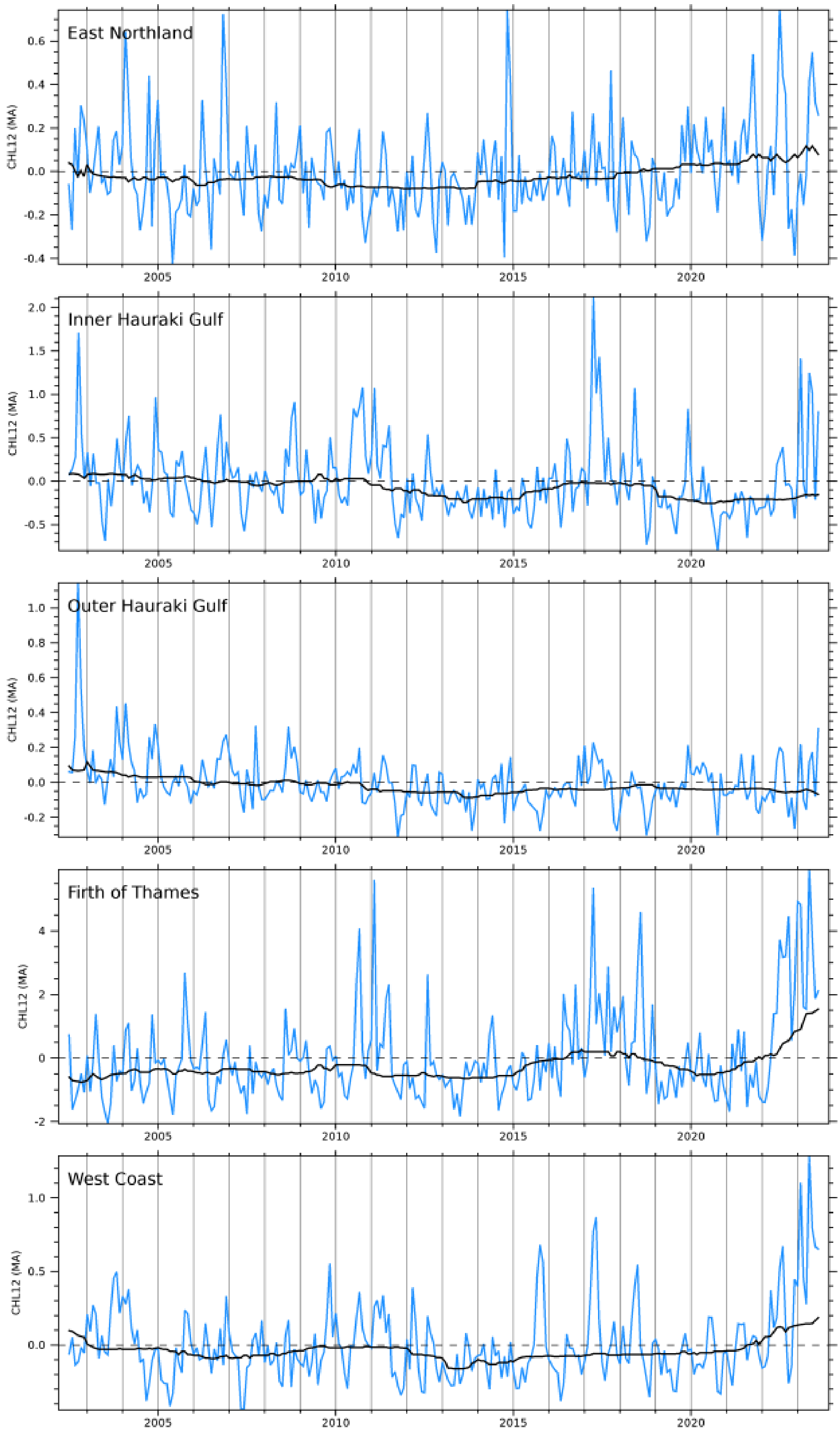



**Figure 11: Monthly chlorophyll-a concentration anomalies (mg $m^{-3}$) from satellite measurements and 4-year smoother (4-year rolling median; black line).**

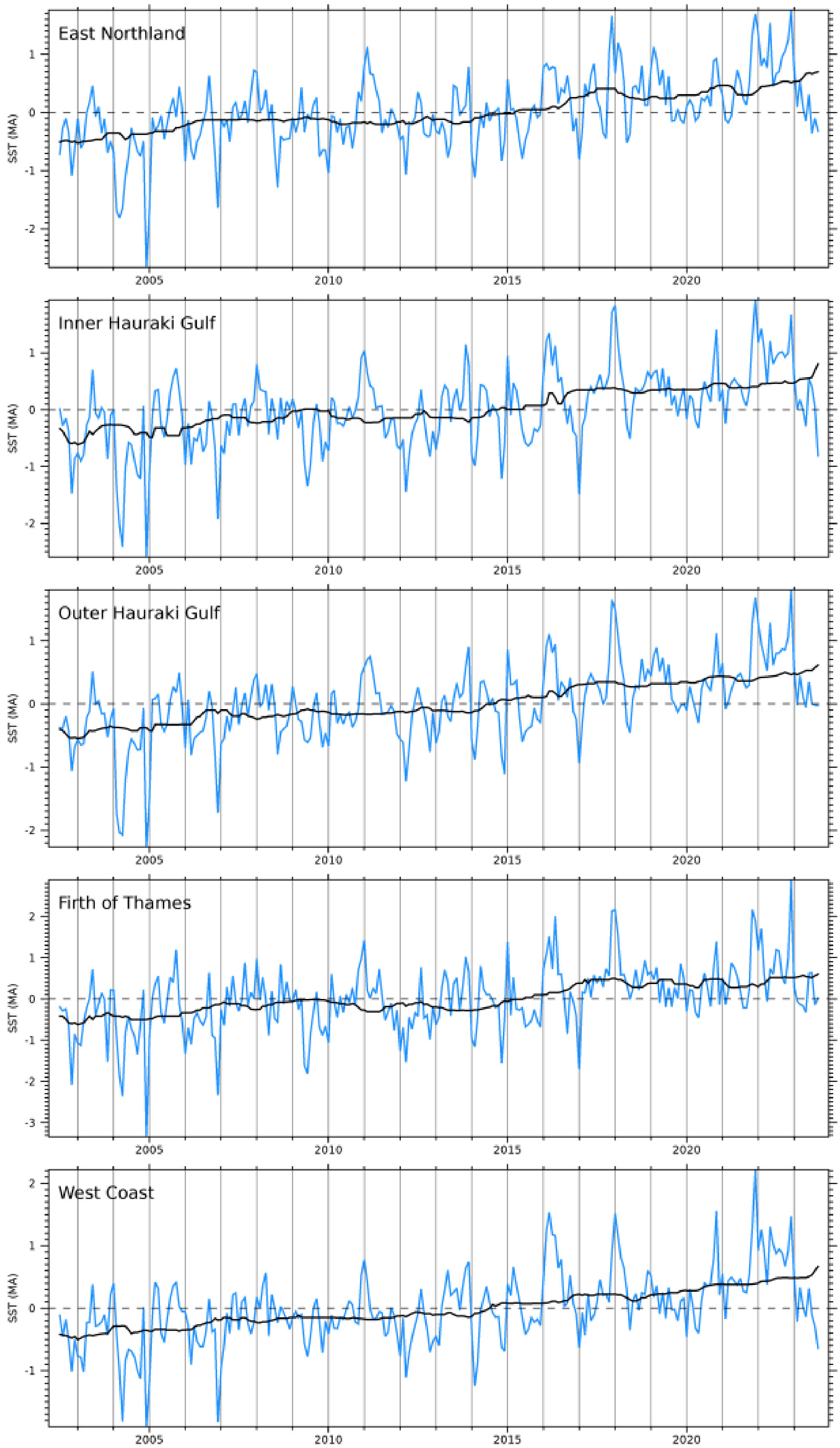



**Figure 12: Monthly sea surface temperature anomalies (°C) from satellite measurements and 4-year smoother (4-year rolling median; black line).**

## Freshwater input

The Piako River and Waihou River collectively contribute the majority of the freshwater input to the Hauraki Gulf. The water level and flow of these rivers was therefore examined as a proxy for the amount of freshwater entering the Hauraki Gulf. Both the Piako and Waihou rivers have a regular annual increase in water level and flow during winter (Figure 13, 14), which would increase the amount of freshwater entering the Hauraki Gulf around this time. Both rivers showed increased water levels and flow from mid-2022, remaining high into 2023, compared with the previous three years. However, this increase appears similar to the high levels and flows in the 2017–2018 years, the previous La Niña event (see Figure 9).

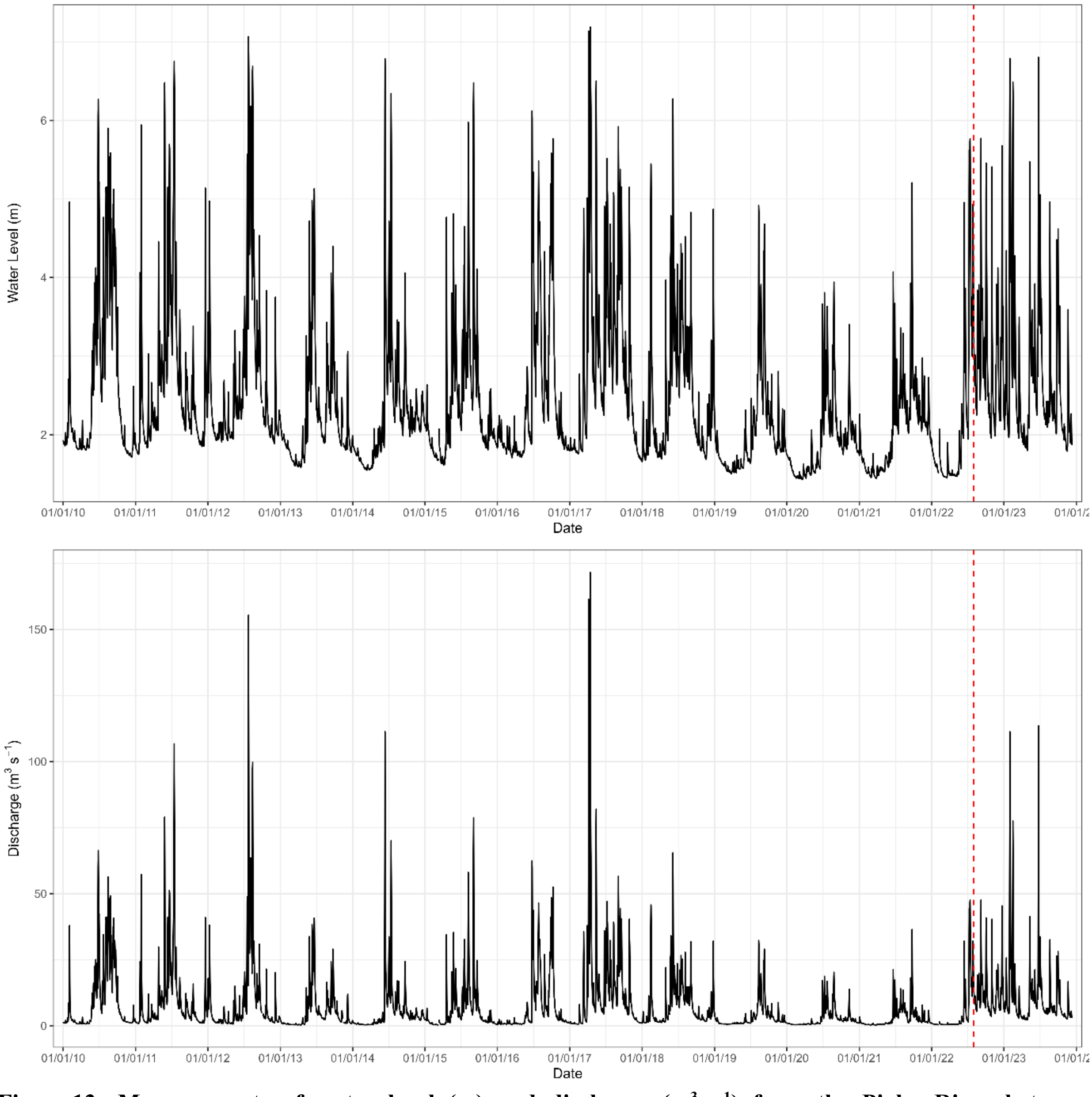



**Figure 13: Measurements of water level (m) and discharge ($m^3$ $s^{-1}$) from the Piako River between 1 January 2010 and 11 December 2023. Measurements are taken from the Paeroa-Tahuna Road measuring station. The red vertical dashed line indicates 1 August 2022, when milky fleshed snapper began to be reported to NIWA. Data provided by Waikato Regional Council.**

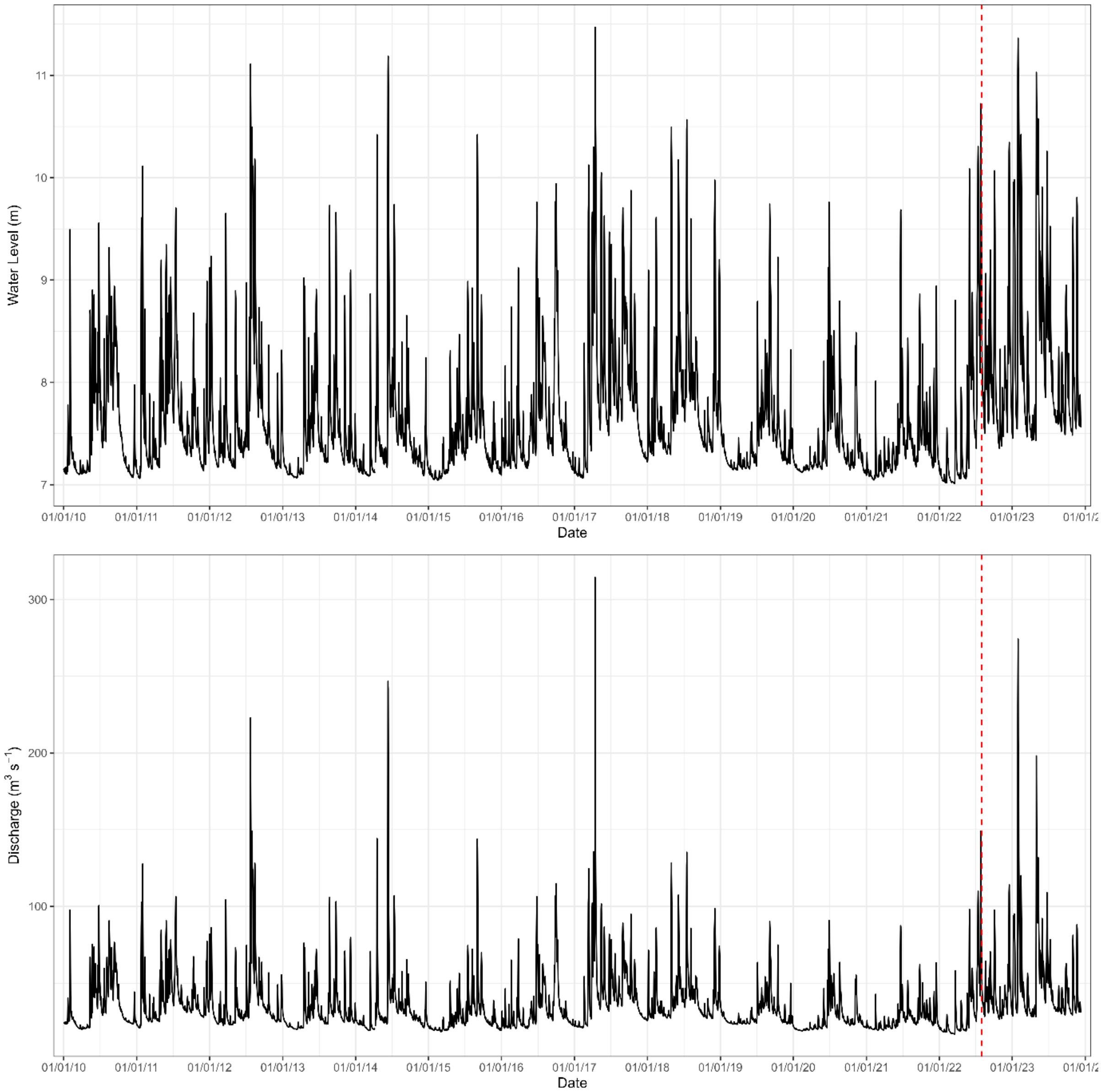



**Figure 14: Measurements of water level (m) and discharge ($m^3\ s^{-1}$) from the Waihou River between 1 January 2010 and 11 December 2023. Measurements are taken from Te Aroha measuring station. The red vertical dashed line indicates 1 August 2022, when milky fleshed snapper began to be reported to NIWA. Data provided by Waikato Regional Council.**

Rainfall in the Hauraki Gulf region and surrounding areas shows a relatively cyclical pattern, with increased rainfall in winter (Figure 15). There was extreme rainfall in the SNA 1 region in early 2023. However, the large increase in reporting of MWF snapper occurred in August 2022, 6 months before this extreme rainfall. Furthermore, looking at the long-term trends across the five representative locations, the 2023 extreme events do not appear to differ markedly from other periods of heavy rainfall during the 13-year period examined. Consequently, it is difficult to identify runoff and/or runoff-related processes as a driving force for MWF syndrome in snapper.

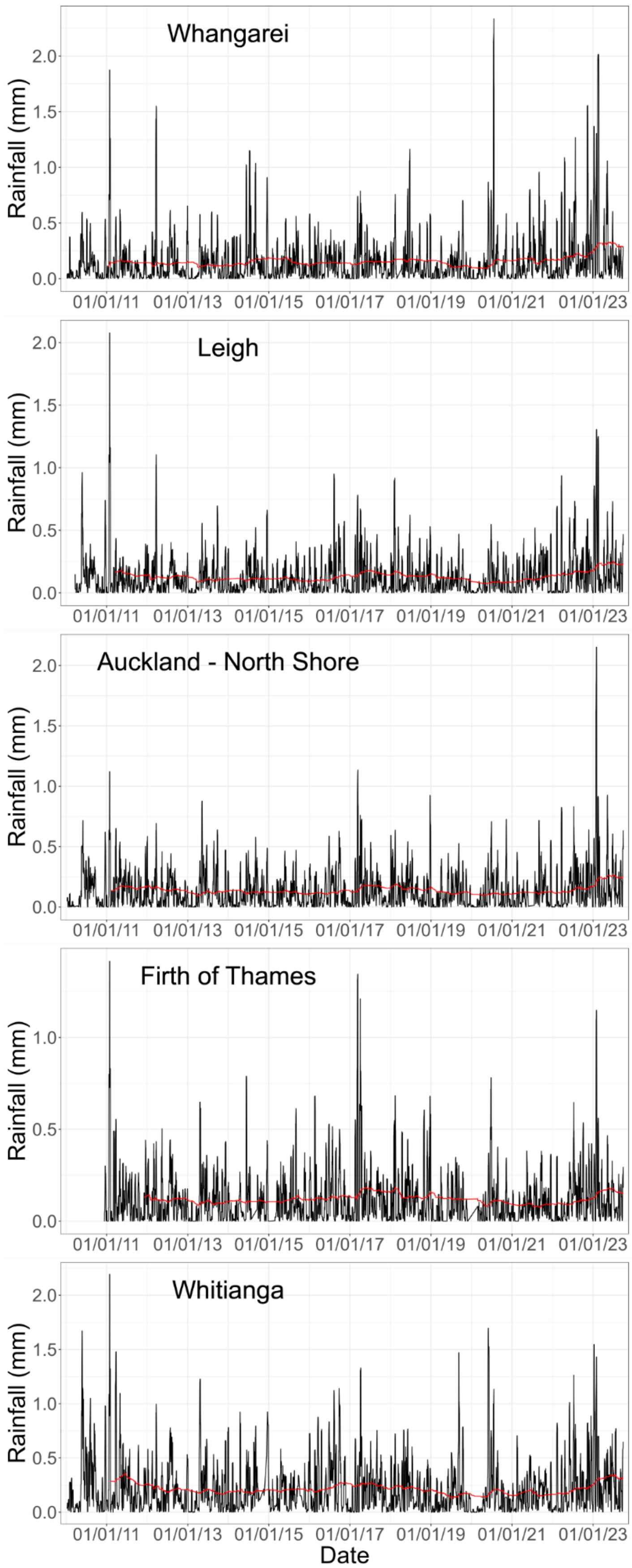



**Figure 15: Average weekly rainfall from locations around the Hauraki Gulf and periphery. The red line indicates the 365-day rolling mean. Data from the NIWA Climate Database.**

Although there was an increase in freshwater entering the Hauraki Gulf in winter 2022 from both river and rainfall/runoff sources, this increased freshwater input is unlikely to be the cause of the increase in abundance of milky fleshed, malnourished snapper; the affected snapper were observed in large quantities around the time of the increased freshwater input but these fish were already in a state of chronic malnutrition. The fish would not become malnourished to this extent quickly, so their malnourished state is likely to be the result of stressors earlier in 2022, when freshwater input to the Hauraki Gulf was at a similar level to previous years.

## Biological recruitment

In recent years, there has been an increase in the recruitment of snapper into the Hauraki Gulf (in the context of the stock assessment, recruitment means fish reaching 25 cm FL). The estimated year class strength (YCS) for the Hauraki Gulf shows a large increase in recent years (Figure 16), which has driven a subsequent increase in snapper biomass, particularly from 2010 onwards (Figure 17). This increased biomass of snapper will ultimately increase the pressure on resources, as demonstrated in a study that found decreased growth in snapper when stock size increased (Morrongiello et al. 2021). Year class strength is not presented for the Bay of Plenty and East Northland populations, but it did not exhibit an increasing trend as seen in the Hauraki Gulf.

The sustained low Chl-a levels and La Niña event in recent years may seem counterintuitive to high larval survival rates. However, studies conducted in northeastern New Zealand have shown that stronger easterly winds result in large quantities of zooplankton, including larval snapper and their food sources (Chang et al. 2003, Zeldis 2004, Zeldis et al. 2005). Furthermore, with reduced upwelling, phytoplankton are distributed over the shelf in shallower waters rather than being predominantly located beyond the edge of the shelf as occurs during upwelling conditions (Chang et al. 2003). This means that during La Niña events with reduced upwelling, food resources are distributed over shallower waters where larval and post-settlement snapper are located.

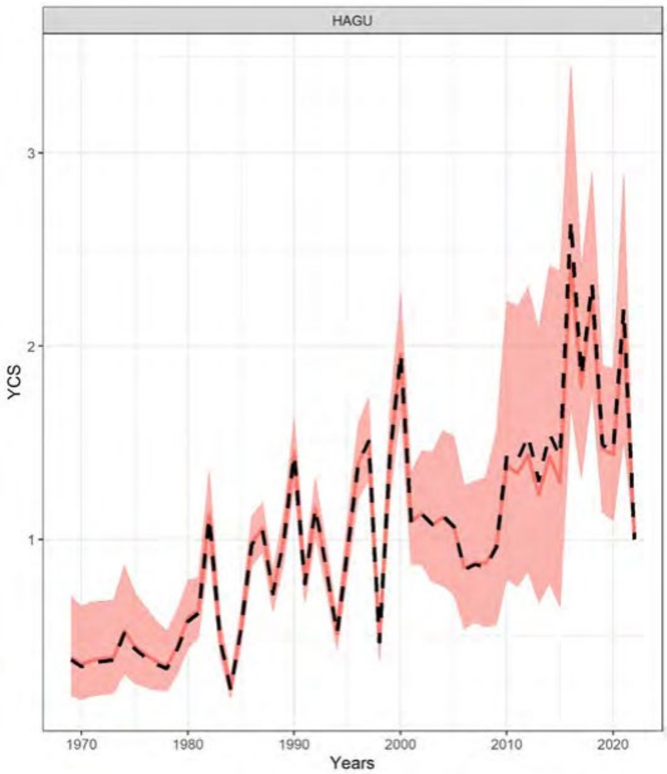



**Figure 16: Snapper estimated year class strengths by year for the Hauraki Gulf. A value of 1 indicates that the year class has the strength predicted by the stock-recruit relationship. Estimates are: Markov chain Monte Carlo median (solid line); 95% confidence intervals (shaded regions); mode of the posterior distribution estimates (dashed line). Plot reproduced with permission from Fisheries New Zealand (2023).**

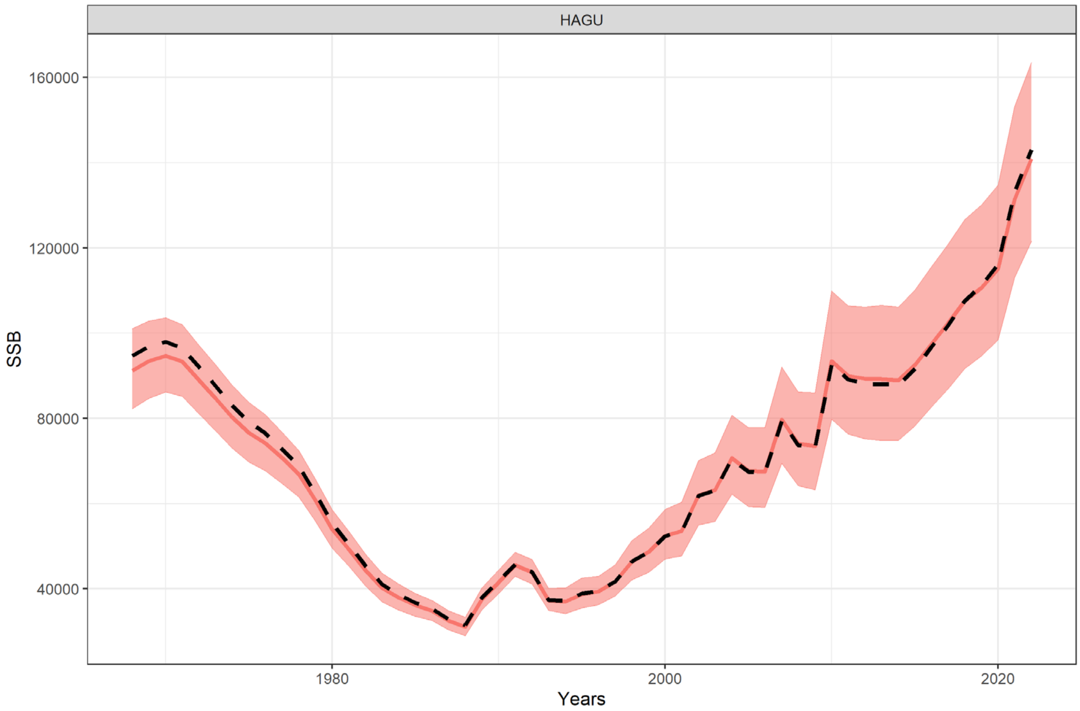



**Figure 17: Snapper estimated Spawning Stock Biomass (tonnes) by year for the Hauraki Gulf. Estimates are: Markov chain Monte Carlo median (solid line); 95% confidence intervals (shaded regions); mode of the posterior distribution estimates (dashed line). Plot reproduced with permission from Fisheries New Zealand (2023).**

## Parasites

In late 2023, there were numerous reports of snapper with abrasions near the head and along the flank. When a snapper is caught and killed, parasites around the abrasion site become opaque and are more easily observed (Figure 18). It seems likely that these parasites cause irritation, and the abrasions are the result of snapper attempting to rid themselves of parasites.

This parasite was identified as a common snapper parasite, likely *Benedenia sekii* (Capsalidae: Monogenea) (K. Hutson, Cawthron, pers. comm). However, *B. sekii* are usually rare and would naturally be overdispersed, with only a few heavily infected fish rather than many individuals being infected. The increased prevalence suggests that snapper are currently more susceptible to the parasite. Increased temperatures can cause changes in parasite-host interactions (Alfonso et al. 2021), so increasing SST combined with snapper in a malnourished state may make MWF snapper more susceptible to infection. Due to the increase in *B. sekii* prevalence being a recent event, we do not have data on whether this parasite is more associated with skinny or MWF snapper, although it seems plausible that MWF snapper may have higher susceptibility to disease. This association should be monitored in the future because it may affect the ability of MWF snapper to recover from their malnourished condition.

**Figure 18: Abrasions observed on the head and flank of snapper (left) and the parasite which appears to be the cause of the irritation, likely *Benedenia sekii* (Capsalidae: Monogenea) (right). Photos supplied by FV *Exenda* and Daniel McLaren.**

## 3.4 Toxins

Because the Hauraki Gulf is the receiving environment for a large catchment that includes agricultural, horticultural, forestry, residential, and industrial land uses, there is a wide variety of chemicals that have potential to be present (albeit, in most cases at low concentrations). There are an estimated 29 000 chemicals approved for use in New Zealand (Environmental Protection Authority 2023), although there is a paucity of publicly available information on the amount of chemicals used (Parliamentary Commissioner for the Environment 2022). While not all of these chemicals will be toxic to marine life, there are a large number that may be—as of 2023, there are fish toxicity data available for 1005 chemicals in the US Environmental Protection Agency's ECOTOX database (Olker et al. 2022)—and the toxicity of many other chemicals has not been assessed.

The potential for toxic effects depends on the concentration of that chemical. However, the concentration (or dose) depends on both the amount of contaminant released into the environment AND how long it persists there (Appendix 2, Table A2.1).

The toxicology and occurrence of toxic metals, legacy persistent organic pollutants, current-use pesticides, and emerging organic contaminants were assessed and subsequently a causality assessment was conducted to gain an understanding of their potential toxic effects in SNA 1 (Appendix 2). These assessments gave greater confidence that legacy pesticides can be ruled out as the cause of MWF syndrome due to monitoring showing that their presence in marine environments is decreasing. Furthermore, livers were collected from snapper with milky flesh during NIWA catch-at-age sampling and were subsequently tested for a range of organochlorine pesticides (OCPs; one group of legacy pesticides). No OCPs tested for were detected in milky fleshed snapper livers, with a detection limit of 0.004 mg $kg^{-1}$ or less. This suggests that there is low likelihood of the adverse effects being due to OCPs, assuming snapper do not have extremely low sensitivity. For the other toxins reviewed, there is a distinct lack of evidence. However, none of the assessed types of toxins are consistent with an effect at the broad spatial scale over which MWF snapper are distributed in SNA 1. At present there is no evidence that toxic substances are the cause of the MWF syndrome.

## 4. CONCLUSIONS

The distribution of MWF snapper within SNA 1 appears to be most concentrated in the inner Hauraki Gulf and in East Northland around Doubtless Bay, with very small numbers observed in the Bay of Plenty (especially east of Tauranga). They appear to be most abundant and widespread in winter, specifically in June to August. Despite interview responses indicating that smaller fish tended to be affected, length frequency analysis suggested that the size distribution of affected fish is similar to the full size range of commercial longline catch. The likely reason for more smaller fish being identified is due to the high prevalence of smaller fish in the Hauraki Gulf where most fishing occurred. MWF snapper were more prevalent in 2023 than 2022, although this finding may be biased by better records in 2023 than 2022. Records also indicate that reef-dwelling snapper are not affected in the same way as snapper over sandy and muddy sediments.

The prolonged La Niña event experienced from mid-2020 to the end of 2022 brought warmer air and water temperatures and stronger easterly winds, likely resulting in less upwelling off the east coast of the North Island. This was reflected in sustained low Chl-a levels in the Hauraki Gulf. However, this was not seen in the Firth of Thames or off the west coast of the North Island where Chl-a has seen a marked increase over the last two years. East Northland also had an increase in Chl-a, but at a lower rate. Sea surface temperatures have been steadily increasing over the last 20 years off both the east and west coasts of the North Island. However, the fact that the Firth of Thames and the west coast—the two systems examined that are physically the most different—have similar trends for Chl-a and TSS suggests that this added pressure is only affecting parts of SNA 1, and alone is not the cause of MWF snapper.

The increased recruitment of snapper to the fishery in the Hauraki Gulf, combined with the above added stress, seems likely to have had an impact on snapper and may be a contributing factor to the prevalence of MWF snapper. The higher biomass of snapper combined with low Chl-a and reduced upwelling may mean that these snapper in the Hauraki Gulf have more limited food resources per capita. Combined with potential increased metabolic rates as a result of the increased SST, there may have been greater strain on resources. There have been anecdotal observations suggesting increased snapper cannibalism (e.g., a number of instances where two snapper are caught on the same hook due to cannibalistic behaviour (Appendix 3)), supporting the suggestion of a strain on food resources. Whilst cannibalism is not unheard of in snapper, it is usually observed when adult snapper consume juveniles (Parsons et al. 2014).

The timing of the increased recruitment, environmental stressors, and changes in the prevalence of MWF snapper are not perfectly aligned, although a lag would be expected between the increase in snapper numbers and potential repercussions such as malnutrition (in the same way that there is a lag in population abundance in the classic lynx and hare predator-prey relationship—the prey (hare) population declines while the predator (lynx) numbers continue to increase, until a point is reached where the predator runs out of prey and also declines, allowing the prey population to begin to recover (Keith 1963)). Therefore, the prevalence of MWF snapper in mid-late 2022 was likely caused by factors that were impacting the system earlier in 2022. However, as noted above, the different trends seen in the Firth of Thames and East Northland, and the fact that these were similar to the west coast where MWF snapper are not as abundant, suggest that the increased recruitment and environmental pressures combined are not the sole root of the problem.

The lack of pathological and bacteriological evidence of parasites in MPI testing, as well as reports of MWF in species other than snapper, makes it unlikely that the cause of the syndrome is parasite-based. The more recent identification of *B. sekii* parasites on snapper is unlikely to be a contributing factor to the cause of MWF snapper. However, it may add stress to recovering malnourished fish should they be infected, so the prevalence of *B. sekii* should be closely monitored to ensure that they do not prevent recovery in these individuals.

A review of the likelihood of toxins being a causative agent gave greater confidence that legacy pesticides can be ruled out, whereas for the other toxins reviewed there is a lack of evidence. However, the long timescales that MWF snapper have been prevalent over and the large spatial scales make it very unlikely that toxins are the cause; the quantity of a toxin needed to cause an effect over such a spatial scale would be unrealistically large, and the longevity that the toxin would need to have for the presence of MWF snapper to still be recorded over a year later makes it unlikely to be a current-use toxin.

There has also been an incursion of the exotic Caulerpa species *Caulerpa brachypus* and *Caulerpa parvifolia* in SNA 1. The invasive species were first detected at Great Barrier Island (Aotea Island) in the Hauraki Gulf in July 2021, with subsequent spread to Great Mercury Island (Ahuahu) and Omākiwi Cove in the Bay of Islands. The toxins in Caulerpa have been shown to have a negative correlation with body condition in the omnivorous Mediterranean white seabream *Diplodus sargus* (Terlizzi et al. 2011), so have the potential to impact species such as snapper that may indirectly consume these toxins. Laboratory testing of exotic Caulerpa in SNA 1 suggests that these species do not have high levels of the more concerning toxic compound caulerpenyne, although its presence cannot be ruled out (J. Scarrott, University of Auckland, pers. comm.). Although the timing of Caulerpa appearing in SNA 1 could be consistent with MWF snapper appearing in large numbers in mid-2022, the wide spatial distribution of MWF snapper makes Caulerpa an unlikely cause; admittedly the full distribution of Caulerpa is likely greater than is currently known, but the quantity needed across the East Northland and Hauraki Gulf regions to have a detrimental impact on snapper at the scale observed seems improbable.

It seems most likely that a combination of factors have caused the MWF syndrome in SNA 1. Long-term low Chl-a concentrations, less upwelling along the east coast, increased snapper biomass in the Hauraki Gulf, and an increased metabolic rate as a result of the increased temperatures are all likely to put strain on the snapper populations. However, when compared to the west coast where MWF snapper are not observed, the environmental correlates examined do not perfectly explain the spatial distribution of MWF snapper.

## 4.1 Impacts

The ongoing prevalence of MWF snapper has already affected commercial fishing in SNA 1, with skippers reporting that they are moving away from the worst affected areas, and has the potential to cause large-scale disruption with nearly all boats leaving the Hauraki Gulf. Decreases in the amount of snapper caught in the Hauraki Gulf will have displacement effects that could undermine snapper populations in East Northland and the Bay of Plenty. Tagging studies of the three biological stocks in SNA 1, East Northland, Hauraki Gulf, and Bay of Plenty have shown that there is limited mixing between the three areas (Fisheries New Zealand 2023). As such, displacement effects from boats leaving the Hauraki Gulf could have a considerable impact on the other two populations.

The presence of MWF snapper comes with a substantial economic impact, with almost 50 t of catch devalued between January and November 2023 (and further catch being rejected due to fish being too thin to fillet). With SNA 1 being the largest snapper fishery in New Zealand, the potential repercussions if the MWF syndrome persists are economically concerning and highlight the importance of monitoring the situation.

The impacts of MWF snapper are of interest to recreational fishers who are concerned about the fishery in SNA 1, particularly in the Hauraki Gulf. The 2017–18 National Panel Survey estimated that 3127 t of snapper were caught recreationally in SNA 1, with more than half of that total coming from the Hauraki Gulf (Fisheries New Zealand 2023), demonstrating the scale of the potential impact should the syndrome persist.

If the syndrome persists, and the affected individuals ultimately die, the syndrome could be a source of substantial additional mortality that would need to be included in future stock assessments. However, it is also possible that the impacts of MWF syndrome were reversed for some snapper in early 2024 while the SOI was variable, with the short El Niño ending in May.

## 5. POTENTIAL RESEARCH

Due to the wide range of impacts of MWF snapper that could substantially affect the SNA 1 fishery, it is important to have ongoing monitoring to assess the situation. Monitoring of commercial SKY grade data is crucial to get comprehensive coverage of the full SNA 1 fishery. Recreational fishers are a useful source of information for the prevalence of MWF snapper, so data from these fishers should continue to be monitored; this includes recreational catches assessed in this report from the Mairangi Bay Fishing Club, data collected by Kai Ika filleting service, Salty Towers filleting database, and seasonal LegaSea public surveys.

To definitively rule out toxins as the cause of MWF syndrome in snapper, further monitoring, measurement, and/or experiments would be needed. Measurements of toxic substances in the water column and in fish with and without MWF syndrome (from locations around SNA 1 and at reference locations) would provide greater understanding of potential direct effects of toxins. Using biomarkers to get a relative measurement of exposure to various toxins could also be valuable to compare fish with and without MWF syndrome. Manipulative experiments would also provide information on how fish respond to toxin exposure or recover from MWF syndrome if affected fish were used. Each approach has benefits and challenges, which are detailed in Appendix 2, Recommendations.

As mentioned above, elevated mortality is expected while the system rebalances, but the proportion of MWF snapper is expected to decrease as affected snapper also recover; recovery is expected with the recent change in environmental conditions, particularly with the recent El Niño event where temperatures are typically comparatively lower than during La Niña events, and, importantly, the increased westerly winds result in upwelling and an increase in nutrients entering the system. However, this needs to be monitored to assess whether MWF snapper are in fact declining in number as expected or whether they are persisting.

The expected mortality as the system rebalances also has an important impact on the stock assessment of SNA 1. Due to this change in the population being an acute change, data need to be captured to assess how big this change is; ongoing monitoring is essential as without data, any hypothetical models may fail to capture such dynamic changes. To accurately assess such dynamic changes in the SNA 1 population, ongoing frequent monitoring is essential to ensure an accurate stock assessment and the long-term success of the SNA 1 fishery.

## 6. ACKNOWLEDGEMENTS

We would like to acknowledge the generous contributions of the many people, organisations, and companies that have made this project possible: Katie Goodwin, Josh Inger, Sydney Curtis, and Sam Woolford of LegaSea for general support and collection of data through Kai Ika and public surveys; Barry Stevens and the Mairangi Bay Fishing Club for collecting data; Salty Towers for sharing their filleting database; Nathan Reid (Moana) and Tom Searle (Lee Fish) for providing data; NIWA staff including Charlotte Bodie, Finn Barry, Maren Howarth, and Helena Armiger for collecting data in the fish sheds; the charter, commercial, and recreational fishers who provided interview responses; Kate Hutson (Cawthron) for advice and identification of parasites; Jessie Scarrott for sharing information on Caulerpa; staff at MPI Biosecurity and Food Safety who tested initial samples; Anjali Pande (MPI) for sharing the Milky White Flesh Syndrome case definition and investigative plan; Marty Deveney (South Australian Research and Development Institute) for advice on toxicology; Andrew Jeffs (University of Auckland) for snapper toxicology testing; Doug Stewart (Waikato Regional Council) for supplying river flow data. Thank you to Marc Griffiths (MPI) for all his work and help with this project, including securing the funding and developing a proposal for the work, as well as valuable input on the project design and report drafts. Thank you to Jeremy McKenzie and Richard O'Driscoll (NIWA) for reviewing earlier drafts of this report. Funding for this project (SEA2022-08) was provided by Fisheries New Zealand.

## 7. REFERENCES

Alfonso, S.; Gesto, M.; Sadoul, B. (2021). Temperature increase and its effects on fish stress physiology in the context of global warming. *Journal of Fish Biology 98 (6)*: 1496–1508.

Allen, H. (2023). Arsenic and mercury in marine sediment: state and preliminary trends in Tāmaki Makaurau / Auckland 2012-2021. *Auckland Council Technical Report 2023/14*. 61 p.

ANZECC; ARMCANZ. (2000). Toxicant default guideline values for aquatic ecosystem protection: DDT. Reproduced in ANZG (2018), Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Australian and New Zealand Governments and Australian state and territory governments, Canberra, ACT, Australia. Retrieved from https://www.waterquality.gov.au/anz-guidelines/guideline-values/default/water-quality-toxicants/toxicants/ddt-2000

ANZG (2018). Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Governments. Retrieved from https://www.waterquality.gov.au/anz-guidelines

ATSDR (2022). Toxicological profile for mercury (draft for public comment). Agency for Toxic Substances and Disease Registry (ATSDR), U.S. Department of Health and Human Services, Public Health Service, Atlanta. Retrieved from https://www.atsdr.cdc.gov/ToxProfiles/tp46.pdf

Balneaves, J.M. (1981). The use of 2,4,5-Trichlorophenoxy acetic acid (2,4,5-T) in forestry in the South Island, New Zealand. *New Zealand Journal of Forestry 26 (2)*: 232–244.

Beckvar, N.; Dillon, T.M.; Read, L.B. (2005). Approaches for linking whole-body fish tissue residues of mercury or DDT to biological effects thresholds. *Environmental Toxicology and Chemistry 24 (8)*: 2094–2105.

Beckvar, N.; Lotufo, G.R. (2011). DDT and other organohalogen pesticides in aquatic organisms. *In*: Beyer, W.N.; Meador, J.P. (Eds.). *Environmental contaminants in biota: interpreting tissue concentrations* (2nd ed.). pp 47–101. CRC, Boca Raton.

Boehnert, S.; Ruiz Soto, S.; Fox, B.R.S.; Yokoyama, Y.; Hebbeln, D. (2020). Historic development of heavy metal contamination into the Firth of Thames, New Zealand. *Geo-Marine Letters 40 (2)*: 149–165.

Bowering, L.R.; McArley, T.J.; Devaux, J.B.L.; Hickey, A.J.R.; Herbert, N.A. (2023). Metabolic resilience of the Australasian snapper (*Chrysophrys auratus*) to marine heatwaves and hypoxia. *Frontiers in Physiology 14*: 1215442.

Breaux, N.; Lebreton, B.; Palmer, T.A.; Guillou, G.; Beseres Pollack, J. (2019). Ecosystem resilience following salinity change in a hypersaline estuary. *Estuarine, Coastal and Shelf Science 225*: 106258.

Buckland, S.J.; Jones, P.D.; Ellis, H.K.; Salter, R.T. (1998). *Organochlorines in New Zealand: Ambient concentrations of selected organochlorines in rivers.* Organochlorines Programme, Ministry for the Environment. 177 p.

Casini, M.; Käll, F.; Hansson, M.; Plikshs, M.; Baranova, T.; Karlsson, O.; Lundström, K.; Neuenfeldt, S.; Gårdmark, A.; Hjelm, J. (2016). Hypoxic areas, density-dependence and food limitation drive the body condition of a heavily exploited marine fish predator. *Royal Society Open Science 3 (10)*: 160416.

Chang, F.H.; Zeldis, J.; Gall, M.; Hall, J. (2003). Seasonal and spatial variation of phytoplankton assemblages, biomass and cell size from spring to summer across the north-eastern New Zealand continental shelf. *Journal of Plankton Research 25 (7)*: 737–758.

Chapman, R.B. (2010). A review of insecticide use on pastures and forage crops in New Zealand. Insect Science Ltd report for AgResearch. 78 p.

Close, M.E. (1996). Survey of pesticides in New Zealand groundwaters, 1994. *New Zealand Journal of Marine and Freshwater Research 30 (4)*: 455–461.

Close, M.E.; Banasiak, L. (2023). National survey of pesticides in groundwater 2022. ESR Client Report CSC23010. Prepared for Regional and Unitary Authorities. 32 p.

Close, M.E.; Humphries, B.; Northcott, G. (2021). Outcomes of the first combined national survey of pesticides and emerging organic contaminants (EOCs) in groundwater in New Zealand 2018. *Science of the Total Environment 754*: 142005.

Cormier, S.M.; Suter, G.W., II; Norton, S.B. (2010). Causal characteristics for ecoepidemiology. *Human and Ecological Risk Assessment: An International Journal 16 (1)*: 53–73.

Crawshaw, J. (2021). Bay of Plenty comprehensive contaminant report 2020. *Bay of Plenty Regional Council, Environmental Publication 2021/07*. 90 p.

Cressey, P.; Pearson, A. (2020). Scoping risk from emerging organic contaminants in New Zealand aquatic food species. *New Zealand Food Safety Technical Paper No: 2020/20*. 99 p.

Crossland, J. (1977). Seasonal reproductive cycle of snapper *Chrysophrys auratus* (Forster) in the Hauraki Gulf. *New Zealand Journal of Marine and Freshwater Research 11 (1)*: 37–60.

Crump, K.L.; Trudeau, V.L. (2009). Mercury-induced reproductive impairment in fish. *Environmental Toxicology and Chemistry 28 (5)*: 895–907.

Depew, D.C.; Basu, N.; Burgess, N.M.; Campbell, L.M.; Devlin, E.W.; Drevnick, P.E.; Hammerschmidt, C.R.; Murphy, C.A.; Sandheinrich, M.B.; Wiener, J.G. (2012). Toxicity of dietary methylmercury to fish: Derivation of ecologically meaningful threshold concentrations. *Environmental Toxicology and Chemistry 31 (7)*: 1536–1547.

Eero, M.; Hjelm, J.; Behrens, J.; Buchmann, K.; Cardinale, M.; Casini, M.; Gasyukov, P.; Holmgren, N.; Horbowy, J.; Hüssy, K.; Kirkegaard, E.; Kornilovs, G.; Krumme, U.; Köster, F.W.; Oeberst, R.; Plikshs, M.; Radtke, K.; Raid, T.; Schmidt, J.; Tomczak, M.T.; Vinther, M.; Zimmermann, C.; Storr-Paulsen, M. (2015). Eastern Baltic cod in distress: biological changes and challenges for stock assessment. *ICES Journal of Marine Science 72 (8)*: 2180–2186.

Environment Protection Authority. (2019). Annual report for the year ended 30 June 2019. 111 p.

Environmental Protection Authority. (2021). Guide to classifying hazardous substances in New Zealand (Version 1.0). 51 p.

Environmental Protection Authority. (2023, January 2023). New Zealand Inventory of Chemicals (NZIoC) Spreadsheet. Retrieved October 2023 from https://www.epa.govt.nz/assets/Uploads/Documents/Hazardous-Substances/New-Zealand-Inventory-of-Chemicals-NZIoC/NZIOC-Spreadsheet-January-2023-Website.xlsx

Fangue, N.A.; Todgham, A.E.; Schulte, P.M. (2021). Thermal biology. *In*: Currie, S.; Evans, D.H. (Eds.). *The physiology of fishes* (5th ed.). pp. 91–103. CRC Press, Boca Raton.

Fenaughty, J.M.; Eastman, J.T.; Sidell, B.D. (2008). Biological implications of low condition factor "axe handle" specimens of the Antarctic toothfish, *Dissostichus mawsoni*, from the Ross Sea. *Antarctic Science 20 (6)*: 537–551.

Fisheries New Zealand (2023). *Fisheries Assessment Plenary, May 2023: stock assessments and stock status*. Compiled by the Fisheries Science Team, Fisheries New Zealand, Wellington, New Zealand. 1904 p.

Fulton, M.H.; Key, P.B.; DeLorenzo, M.E. (2014). Insecticide toxicity in fish. *In*: Tierney, K.B.; Farrell, A.P.; Brauner, C.J. (Eds.). *Organic chemical toxicology of fishes* (Vol. 33). Pp. 309–368. Elsevier Academic Press, London.

Godfriaux, B.L. (1969). Food of predatory demersal fish in Hauraki Gulf. 1. Food and feeding habitats of snapper. *New Zealand Journal of Marine and Freshwater Research 3 (4)*: 518–544.

Godfriaux, B.L. (1974). Feeding relationships between tarakihi and snapper in western Bay of Plenty, New Zealand. *New Zealand Journal of Marine and Freshwater Research 8 (4)*: 589–609.

Golder Associates (NZ) Ltd. (2018). Ports of Auckland: Assessment of sediment quality and biosecurity for dumping of dredged sediment from the Port of Auckland at the Cuvier Dump Site. *1779496-002-R-Rev0*. 230 p.

Gous, S. (2003). Overview of the use of agri-chemicals and alternatives to control competing vegetation in forestry. 47 p. https://fgr.nz/documents/download/5954

Hageman, K.J.; Aebig, C.H.F.; Luong, K.H.; Kaserzon, S.L.; Wong, C.S.; Reeks, T.; Greenwood, M.; Macaulay, S.; Matthaei, C.D. (2019). Current-use pesticides in New Zealand streams: Comparing results from grab samples and three types of passive samplers. *Environmental Pollution 254*: 112973.

Hollender, J.; Schymanski, E.L.; Ahrens, L.; Alygizakis, N.; Béen, F.; Bijlsma, L.; Brunner, A.M.; Celma, A.; Fildier, A.; Fu, Q.; Gago-Ferrero, P.; Gil-Solsona, R.; Haglund, P.; Hansen, M.; Kaserzon, S.; Kruve, A.; Lamoree, M.; Margoum, C.; Meijer, J.; Merel, S.; Rauert, C.; Rostkowski, P.; Samanipour, S.; Schulze, B.; Schulze, T.; Singh, R.R.; Slobodnik, J.; Steininger-Mairinger, T.; Thomaidis, N.S.; Togola, A.; Vorkamp, K.; Vulliet, E.; Zhu, L.; Krauss, M. (2023). NORMAN guidance on suspect and non-target screening in environmental monitoring. *Environmental Sciences Europe 35*: 75.

Jobling, S.; Nolan, M.; Tyler, C.R.; Brighty, G.; Sumpter, J.P. (1998). Widespread sexual disruption in wild fish. *Environmental Science and Technology 32 (17)*: 2498–2506.

Keith, L.B. (1963). *Wildlife's ten-year cycle.* University of Wisconsin, Madison. 201 p.

Kiessling, A.; Lindahl-Kiessling, K.; Kiessling, K.H. (2004). Energy utilization and metabolism in spawning migrating Early Stuart sockeye salmon (*Oncorhynchus nerka*): the migratory paradox. *Canadian Journal of Fisheries and Aquatic Sciences 61 (3)*: 452–465.

Kim, N. (2007). Trace elements in sediments of the lower eastern coast of the Firth of Thames. *Environment Waikato Technical Report 2007/08*. 72 p.

Kimbrough, K.L.; Johnson, W.E.; Lauenstein, G.G.; Christensen, J.D.; Apeti, D.A. (2008). An assessment of two decades of contaminant monitoring in the nation's coastal zone. *NOAA Technical Memorandum NOS NCCOS 74*. 105 p.

Kroon, F.; Streten, C.; Harries, S. (2017). A protocol for identifying suitable biomarkers to assess fish health: A systematic review. *PLOS ONE 12 (4)*: e0174762.

Lambert, Y.; Dutil, J.D. (2000). Energetic consequences of reproduction in Atlantic cod (*Gadus morhua*) in relation to spawning level of somatic energy reserves. *Canadian Journal of Fisheries and Aquatic Sciences 57 (4)*: 815–825.

Limburg, K.E.; Casini, M. (2019). Otolith chemistry indicates recent worsened Baltic cod condition is linked to hypoxia exposure. *Biology Letters 15 (12)*: 20190352.

Little, A.G.; Loughland, I.; Seebacher, F. (2020). What do warming waters mean for fish physiology and fisheries? *Journal of Fish Biology 97 (2)*: 328–340.

Manktelow, D.; Stevens, D.; Walker, J.; Gurnsey, S.; Park, N.M.; Zabkiewicz, J.; Teulon, D.; Rahman, A. (2005). Trends in pesticide use in New Zealand: 2004. Report to the Ministry for the Environment, Project SMF4193. 78 p.

Meyers, T.; Burton, T.; Bentz, C.; Ferguson, J.; Stewart, D.; Starkey, N. (2019). Diseases of wild and cultured fishes in Alaska. Alaska Department of Fish and Game, Anchorage, AK. 128 p.

Mills, G. (2014). Marine sediment contaminant monitoring: organic contaminant data review 2003-2010. *Auckland Council Technical Report TR2014/001*. 41 p.

Mills, G.; Allen, H. (2021). Marine sediment contaminant state and trends in Tāmaki Makaurau / Auckland 2004-2019. State of the environment reporting. *Auckland Council Technical Report TR2021/10*. 111 p.

Milne, J.R. (2010). Wellington Harbour marine sediment quality investigation: Supplementary report. *GW/EMI-T-10/76*. 26 p.

Ministry for the Environment (1998). Reporting on persistent organochlorines in New Zealand. Organochlorines Programme, Ministry for the Environment. 36 p.

Morcillo, P.; Esteban, M.A.; Cuesta, A. (2017). Mercury and its toxic effects on fish. *AIMS Environmental Science 4 (3)*: 386-402.

Moreau, M.; Hadfield, J.; Hughey, J.; Sanders, F.; Lapworth, D.J.; White, D.; Civil, W. (2019). A baseline assessment of emerging organic contaminants in New Zealand groundwater. *Science of the Total Environment 686*: 425–439.

Morrongiello, J.R.; Horn, P.L.; Ó Maolagáin, C.; Sutton, P.J.H. (2021). Synergistic effects of harvest and climate drive synchronous somatic growth within key New Zealand fisheries. *Global Change Biology 27 (7)*: 1470–1484.

Oliver, M.D. (2014). Wellington Harbour subtidal sediment quality monitoring: Results from the 2011 survey. *GW/ESCI-T-14/2*. 68 p.

Oliver, M.D.; Conwell, C. (2014). Te Awarua-o-Porirua Harbour subtidal sediment quality monitoring: Results from the 2010 survey. *GW/ESCI-T-14/110*. 125 p.

Olker, J.H.; Elonen, C.M.; Pilli, A.; Anderson, A.; Kinziger, B.; Erickson, S.; Skopinski, M.; Pomplun, A.; LaLone, C.A.; Russom, C.L.; Hoff, D. (2022). The ECOTOXicology Knowledgebase: A curated database of ecologically relevant toxicity tests to support environmental research and risk assessment. *Environmental Toxicology and Chemistry 41 (6)*: 1520–1539.

Olsen, Z.T. (2014). Potential impacts of extreme salinity and surface temperature events on population dynamics of black drum, *Pogonias cromis*, in the upper Laguna Madre, Texas. *Gulf of Mexico Science 32 (1-2)*: 60–68 6.

Olsen, Z.T. (2016). Emaciated black drum (*Pogonias cromis*) in the upper Laguna Madre, Texas: tracking the recovery of the population over two years. *Texas Journal of Science 68 (4)*: 79–90.

Ortega, S.; Lindberg, S.K.; Olsen, S.H.; Anderssen, K.E.; Heia, K. (2023). Early identification of mushy Halibut syndrome with hyperspectral image analysis. *Lwt-Food Science and Technology 176*: 114559.

Park, S. (2014). Bay of Plenty marine sediment contaminants survey 2012. *Bay of Plenty Regional Council, Environmental Publication 2014/03*. 39 p.

Parliamentary Commissioner for the Environment. (2022). Knowing what's out there: Regulating the environmental fate of chemicals. 180 p.

Parsons, D.M.; Morrison, M.A.; Gillanders, B.M.; Clements, K.D.; Bury, S.J.; Bian, R.; Spong, K.T. (2016). Variation in morphology and life-history strategy of an exploited sparid fish. *Marine and Freshwater Research 67 (10)*: 1434–1444.

Parsons, D.M.; Sim-Smith, C.J.; Cryer, M.; Francis, M.P.; Hartill, B.; Jones, E.G.; Le Port, A.; Lowe, M.; McKenzie, J.; Morrison, M.; Paul, L.J.; Radford, C.; Ross, P.M.; Spong, K.T.; Trnski, T.; Usmar, N.; Walsh, C.; Zeldis, J. (2014). Snapper (*Chrysophrys auratus*): a review of life history and key vulnerabilities in New Zealand. *New Zealand Journal of Marine and Freshwater Research 48 (2)*: 256–283.

RNO. (2000). Surveillance du Milieu Marin. Travaux du RNO. Edition 2000. *Ifremer et Ministère de l'Aménagement du Territoire et de l'Environnement, Bulletin RNO 2000*. 32 p.

RNO. (2006). Surveillance du Milieu Marin. Travaux du RNO. Edition 2006. *Ifremer et Ministère de l'Ecologie et du Développement Durable, Bulletin RNO 2006*. 51 p.

Roberts, R.D.; Forrest, B.M. (1999). Minimal impact from long-term dredge spoil disposal at a dispersive site in Tasman Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research 33 (4)*: 623–633.

Scott, S.G.; Pankhurst, N.W. (1992). Interannual variation in the reproductive cycle of the New Zealand snapper *Pagrus auratus* (Bloch & Schneider) (Sparidae). *Journal of Fish Biology 41 (5)*: 685–696.

Shears, N.; Evans, J.; Atkins, J. (2023). Long-term sea surface temperature measurements from the Leigh Marine Laboratory, northern New Zealand. https://doi.org/10.17608/k6.auckland.24773262.v2

Solomon, K.R.; Dalhoff, K.; Volz, D.; Van Der Kraak, G. (2013). Effects of herbicides on fish. *In*: Tierney, K.B.; Farrell, A.P.; Brauner, C.J. (Eds.). *Fish Physiology Volume 33*. pp. 369–409. Academic Press.

Sorensen, P.G.; Milne, J.R. (2009). Porirua Harbour targeted intertidal sediment quality assessment. *GW/EMI-T-09/136*. 71 p.

Stewart, M.; Cameron, M.; McMurtry, M.; Sander, S.G.; Benedict, B.; Graham, L.; Hosie, M.; Green, T. (2016). Development of passive sampling devices for bioavailable contaminants of current and emerging concern: Waitemata Harbour case study. *New Zealand Journal of Marine and Freshwater Research 50 (4)*: 526–548.

Stewart, M.; Gadd, J.; Ballantine, D.; Olsen, G. (2013). Shellfish contaminant monitoring programme: status and trends analysis 1987-2011. *Auckland Council Technical Report TR2013/054*. 187 p.

Strong, J. (2005). Antifoulant and trace metal contamination of sediments from the Napier Inner Harbour. *EMI0511, HBRC Plan Number: 3687*. 30 p.

Terlizzi, A.; Felline, S.; Lionetto, M.G.; Caricato, R.; Perfetti, V.; Cutignano, A.; Mollo, E. (2011). Detrimental physiological effects of the invasive alga *Caulerpa racemosa* on the Mediterranean white seabream *Diplodus sargus*. *Aquatic Biology 12 (2)*: 109–117.

Trenberth, K.E. (1976). Fluctuations and trends in indices of the southern hemispheric circulation. *Quarterly Journal of the Royal Meteorological Society 102 (431)*:65–75.

U.S. Environmental Protection Agency. (2009). The national study of chemical residues in lake fish tissue. *EPA-823-R-09-006*.

U.S. Environmental Protection Agency. (2017, 9 January 2024). Causal Analysis/Diagnosis Decision Information System (CADDIS). Office of Research and Development. www.epa.gov/caddis

U.S. Environmental Protection Agency. (2023, 9 June 2023). About the TSCA Chemical Substance Inventory. Retrieved 16 October 2023 from https://www.epa.gov/tsca-inventory/about-tsca-chemical-substance-inventory

Uphoff, J.H. (2003). Predator-prey analysis of striped bass and Atlantic menhaden in upper Chesapeake Bay. *Fisheries Management and Ecology 10 (5)*: 313–322.

Usmar, N.R. (2012). Ontogenetic diet shifts in snapper (*Pagrus auratus*: Sparidae) within a New Zealand estuary. *New Zealand Journal of Marine and Freshwater Research 46 (1)*: 31–46.

Van Veld, P.A.; Nacci, D.E. (2008). Toxicity resistance. *In*: Di Giulio, R.; Hinton, D.E. (Eds.). *The toxicology of fishes*. pp 597–644. CRC Press, Boca Raton.

Vogue, P.A.; Kerle, E.A.; Jenkins, J.J. (1994). OSU Extension Pesticide Properties Database http://npic.orst.edu/ingred/ppdmove.htm

Volkoff, H.; Rønnestad, I. (2020). Effects of temperature on feeding and digestive processes in fish. *Temperature 7 (4)*: 307–320.

Walsh, C.; Parsons, D.; Bian, R.; McKenzie, J.; Armiger, H.; Taylor, R.; Evans, O.; Buckthought, D.; Smith, M.; Spong, K. (2022). Age composition of commercial snapper landings in SNA 1 and SNA 2, 2019–20. *New Zealand Fisheries Assessment Report 2022/24*. 136 p.

Wang, K.Y.; Peng, C.Z.; Huang, J.L.; Huang, Y.D.; Jin, M.C.; Geng, Y. (2013). The pathology of selenium deficiency in *Cyprinus carpio* L. *Journal of Fish Diseases 36 (7)*: 609–615.

Wilson, N.; Horrocks, J. (2008). Lessons from the removal of lead from gasoline for controlling other environmental pollutants: A case study from New Zealand. *Environmental Health 7*: 1.

Zador, S.; Yasumiishi, E. (2017). Ecosystem considerations 2017: Status of the Gulf of Alaska marine ecosystem. North Pacific Fishery Management Council Gulf of Alaska SAFE. 213 p.

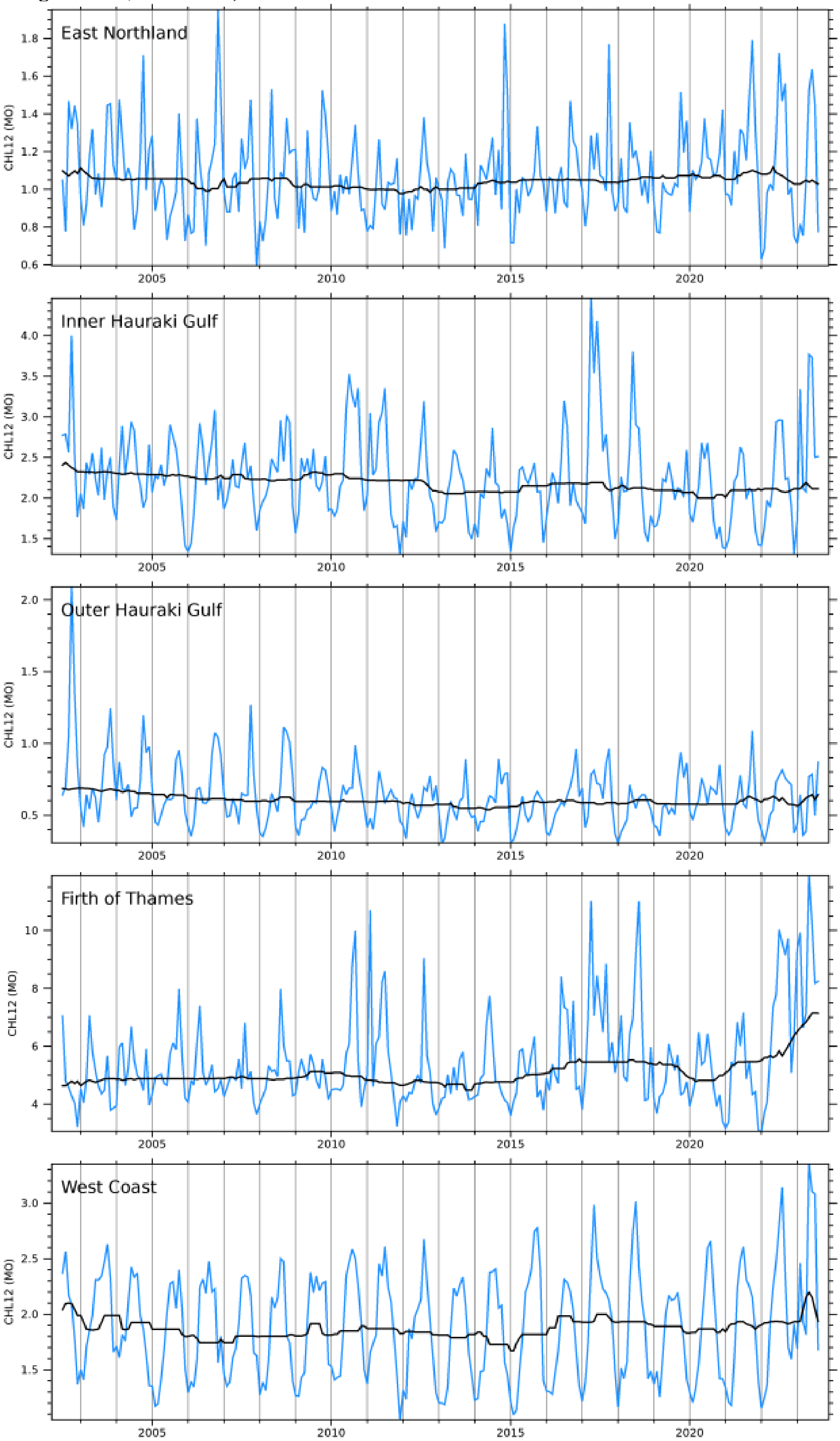
Zeldis, J.R. (2004). New and remineralised nutrient supply and ecosystem metabolism on the northeastern New Zealand continental shelf. *Continental Shelf Research 24 (4)*: 563–581.

Zeldis, J.R.; Oldman, J.; Ballara, S.L.; Richards, L.A. (2005). Physical fluxes, pelagic ecosystem structure, and larval fish survival in Hauraki Gulf, New Zealand. *Canadian Journal of Fisheries and Aquatic Sciences 62 (3)*: 593–610.

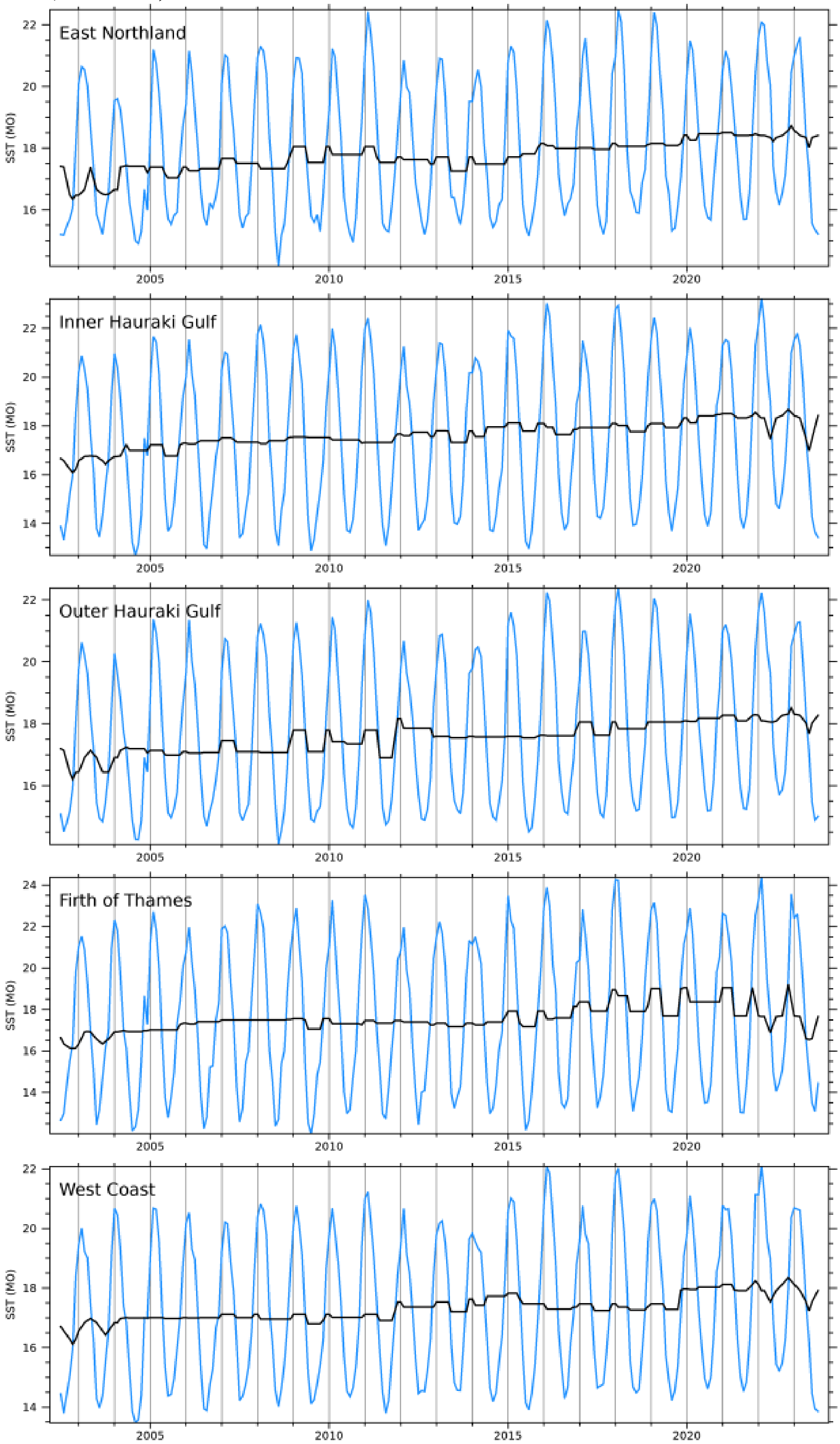
Zhang, F.; Teng, Z.; Wang, L.; Wang, L.; Huang, T.; Zhang, X. (2022). Dietary selenium deficiency and excess accelerate ubiquitin-mediated protein degradation in the muscle of rainbow trout (*Oncorhynchus mykiss*) via Akt/FoxO3a and NF-κB signaling pathways. *Biological Trace Element Research 200 (3)*: 1361–1375.

## APPENDIX 1: Satellite data

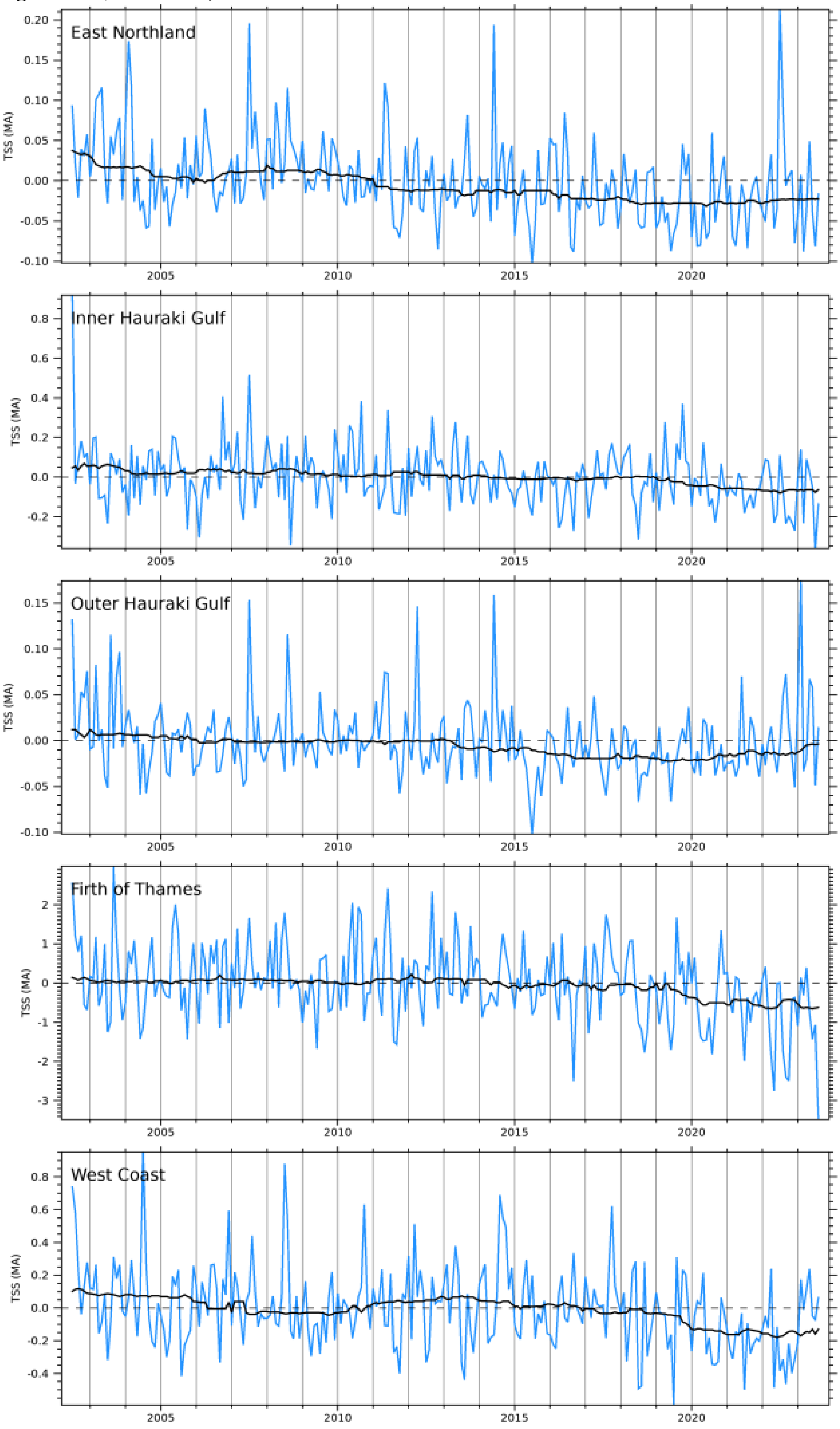
**Average monthly chlorophyll-a concentration (mg m$^{-3}$) from satellite measurements with 4-year smoother (4-year rolling median; black line).**



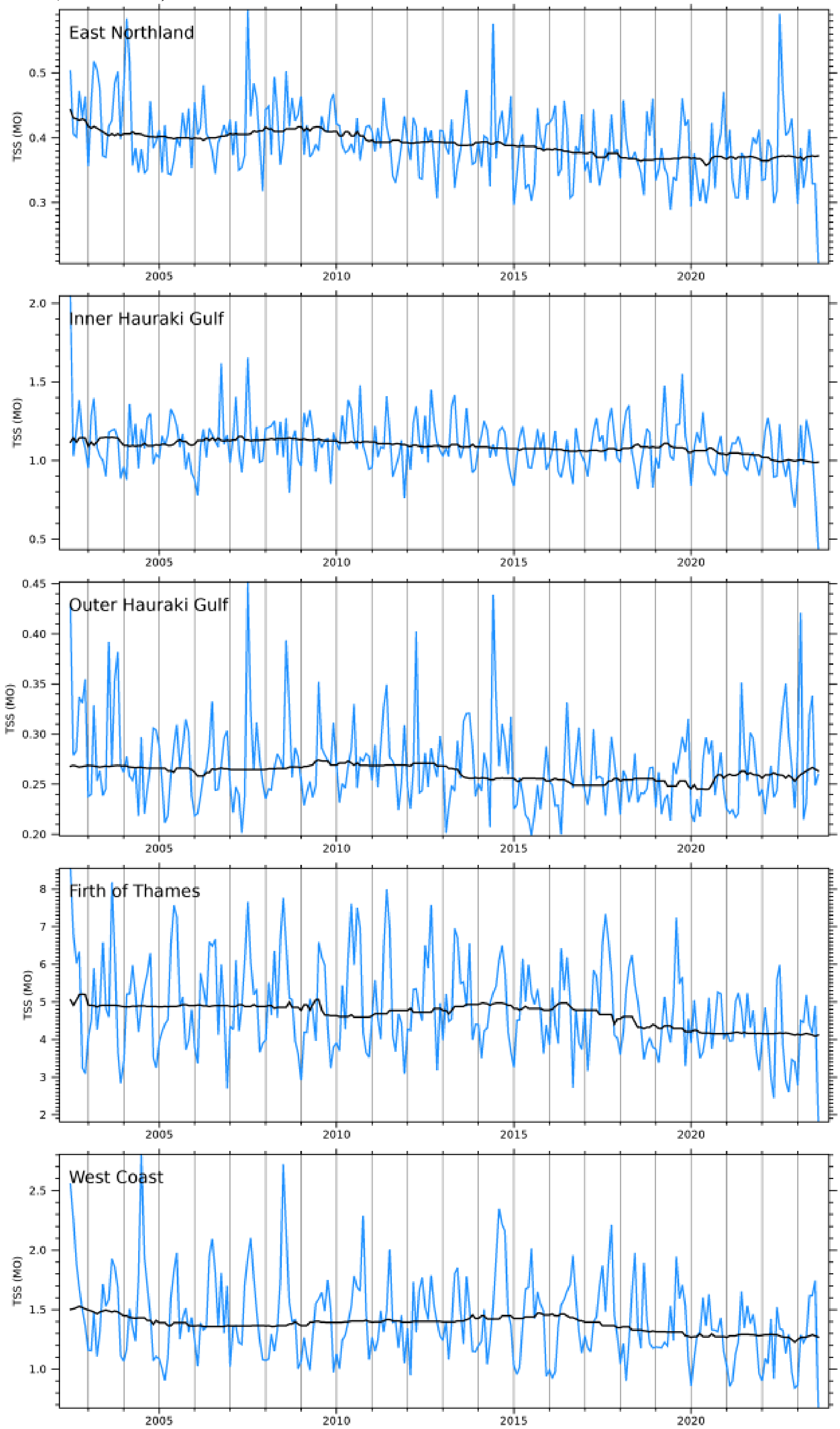
**Average monthly sea surface temperature (°C) from satellite measurements and 4-year smoother (4-year rolling median; black line).**



**Monthly total suspended solids anomalies (g m$^{-3}$) from satellite measurements and 4-year smoother (4-year rolling median; black line).**



**Average monthly total suspended solids (g m$^{-3}$) from satellite measurements and 4-year smoother (4-year rolling median; black line).**



## APPENDIX 2: Review of potential toxicity effects

### Background information

There are over 86 000 chemicals listed on the US Toxic Substances Control Act (TSCA) Inventory (U.S. Environmental Protection Agency 2023) and of the estimated 29 000 chemicals approved for use in New Zealand, a recent report by the Parliamentary Commissioner for the Environment emphasises the lack of data on chemical contaminants in New Zealand, including pesticides (Parliamentary Commissioner for the Environment 2022). The New Zealand EPA has also previously identified chemical use and release as a knowledge gap, stating that "little is currently known about New Zealand's chemical landscape or the impact it has on human and environmental health" (Environment Protection Authority 2019).

The potential for toxic effects depends on the concentration of that chemical, which in turn depends on both the amount of contaminant released and how long it persists in the environment (Table A2.1). Chemicals that are persistent in the environment (resistant to degradation mediated either by microbial or abiotic (photolysis, hydrolysis) processes) can accumulate to higher concentrations over time and eventually reach toxic concentrations. This is particularly an issue for bioaccumulative compounds (compounds that accumulate within biota), which build up in lipids, particularly in larger, long-lived organisms. Many chemicals classed as persistent, bioaccumulative, and toxic (PBT), such as the synthetic pesticide dichlorodiphenyltrichloroethane (DDT), were strictly regulated once evidence surfaced of their accumulation in biota and toxic effects in fish, birds, and mammals (highlighted in Rachel Carson's 1962 book *Silent Spring*).

However, even compounds that degrade rapidly can result in toxic effects if constantly released to the environment at concentrations that can cause effects. An example of this is the discovery of intersex fish in English rivers, due to the continual release of steroid oestrogens and alkylphenol surfactants in wastewater discharges (Jobling et al. 1998).

**Table A2.1: Concepts used in assessing potential effects from contaminants.**

| | Description | Examples |
|---|---|---|
| Persistent | Refers to chemicals that degrade very slowly, for example lasting years in the environment | DDT has a half-life of over 30 years in soil |
| Bioaccumulative | Compounds that build-up within biological organisms, typically non-polar, lipophilic (fat-loving) compounds. This includes compounds that biomagnify – increasing in concentration up the food chain. | Mercury accumulates in biota and is found at highest concentrations in fish that eat other fish |
| Toxic | The capacity to cause adverse effects, which may be acute or chronic and may include carcinogenicity or genotoxicity. Some trace elements may be essential at very low concentrations but cause toxicity at higher concentrations. | Lead is a well-known toxic metal and is also carcinogenic with no safe concentration. |
| Mobile | Refers to the ability of a chemical to move, generally from the place of application (e.g., soils) to a location where adverse effects may occur (e.g., groundwater used as drinking water). | Atrazine is a pesticide that is highly mobile and easily leaches through soil into groundwater and surface waters. |

## Toxic metals

### Introduction

Metals are natural elements but can increase in concentration in coastal and marine waters due to anthropogenic processes, such as mining and release of metal-enriched waters. Some metals, such as zinc, are essential elements, necessary for biological processes at low concentration, but result in toxic effects when present at higher concentrations and in soluble forms, i.e., not as a solid metal. Other metals, such as lead and cadmium, are not essential and any amount can contribute to negative effects.

Because metals are elements, they do not degrade over time. They can accumulate in "environmental sinks" such as sediment, and depending on the metal and organism, can bioaccumulate (build up in biota) making them easier to detect than many other toxins. Mercury can form an organic compound called methylmercury which is highly bioaccumulative and biomagnifies – increasing in concentration up the food chain. Larger, long-lived marine fish and mammals typically have the highest concentrations of mercury (ATSDR 2022).

### Toxicology and mode of action

Metals can result in both acute (based on short-term, higher-level exposure) and chronic toxicity (from long-term, lower-level exposure), with either causing mortality, reduced growth, or reduced reproduction, depending on the specific metal and organism. For some metals (cadmium, copper, zinc) the mode of toxicity is due to the metal mimicking essential nutrients, such as sodium and calcium, and disrupting the usual ionic processes and pathways. This can cause effects on many organs and result in reduced growth and mortality. For mercury, the primary target tissues are the central nervous system (CNS) and the kidney, triggering loss of appetite, brain lesions, cataracts, abnormal motor coordination, and behavioural changes (Morcillo et al. 2017). Reduced weight and emaciation have also been reported for exposed fish (see review by Depew et al. 2012).

Metals can be regulated by fish to a certain extent – most metals will bind to the metallothionein proteins, which reduces internal availability of the metals (Van Veld & Nacci 2008). Measurement of metallothionein is a useful biomarker of exposure to metals, although measurement of metals within biota (gills, liver, flesh) would also indicate whether metals are accumulating to concentrations that could be causing sub-lethal effects.

### Occurrence

Metals are regularly monitored in environmental matrices, by many councils around New Zealand. This includes monitoring of key metals (typically copper, zinc and in some cases lead) in the water of streams and rivers, in estuarine/harbour sediments, and sometimes in stream bed sediments. Monitoring of Auckland estuarine sediments demonstrates that lead and possibly copper are decreasing in concentration at most sites, whereas zinc is increasing at most sites (Figure A2.1) (Mills & Allen 2021). As metals do not degrade, the decreasing concentrations are likely attributable to influx of less contaminated sediment that buries more contaminated sediments. For lead, this is due to the lower concentrations in stormwater since tetraethyl lead was restricted from 1986 (see Wilson & Horrocks (2008) for a detailed timeline of regulation) and then fully removed from petrol in 1996. It is unclear why copper concentrations would have decreased in estuarine sediments as a key source (brake pad wear) is expected to be increasing. However, the sources of copper are less well-known compared to lead and zinc and changes in stormwater treatment and industrial discharges may contribute. Arsenic and mercury have also been regularly monitored since 2012 (Mills & Allen 2021) and, though there are no clear trends over time, there are some spatial patterns, with mercury typically higher at sites that are also high in copper, lead, and zinc; e.g., sites affected by stormwater and wastewater discharges (Allen 2023). Mercury is generally increasing world-wide, as emissions (including atmospheric emissions) increase and long-range transport delivers mercury to locations far from the sources, resulting in widespread contamination (Crump & Trudeau 2009).

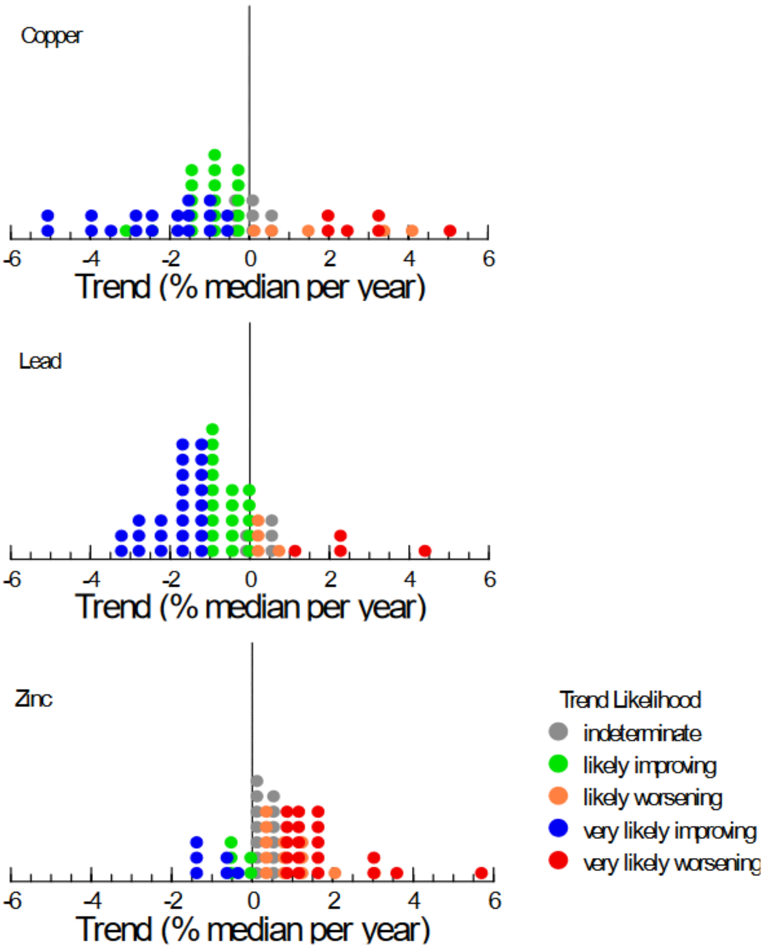



**Figure A2.1: Distribution of trends in copper, lead, and zinc from 2004 to 2019 from 56 monitoring sites. Figure from Mills & Allen (2021), sourced from Auckland Council and subject to copyright.**

Lead and zinc have also been measured at elevated concentrations in the sediments of the Firth of Thames (Boehnert et al. 2020) and attributed to the historic metal mining in the Coromandel. Though there was variation between sites, the highest concentrations measured were not in surface sediments, but at depths of 10–60 cm below the surface, where concentrations were at times above the default guideline values provided by ANZG (2018). The authors (Boehnert et al. 2020) concluded that these higher concentrations in the subsurface sediments pose potential threats if resuspended.

Lead, zinc, and other trace elements (arsenic, cadmium, copper, chromium, mercury, and nickel) were measured in surface sediments of the Firth of Thames in a study by Waikato Regional Council (Kim 2007). This work indicated that historical mining in the Coromandel, including in the Ohinemuri catchment, may have been a significant source of cadmium, lead, and zinc to sediments; agricultural treatments are ongoing sources of cadmium and zinc; elevated copper and arsenic may be due to mining and mineralogy of the coastal Coromandel Peninsula (Kim 2007). Mercury, which was the most enriched of the eight elements monitored (about seven times higher in sediments from the Firth of Thames than at reference sites), was at highest concentrations in the Piako River (and at a hot-spot around a landfill) and was thought to originate from the wetlands and peatlands of the Hauraki Plains (Kim 2007).

## Legacy persistent organic pollutants

### Introduction

A key group of toxic substances is the group known as persistent organic pollutants (POPs). This group includes man-made organic compounds which are resistant to degradation and therefore accumulate in the environment, either within abiotic matrices such as sediment or within biota such as fish and shellfish. Organochlorine pesticides (OCPs), such as DDT, are considered POPs. These pesticides, including DDT, dieldrin, endrin, aldrin, lindane, and PCP were manufactured and heavily used in New Zealand from the 1940s onwards as sheep dips, to control grass grub on pasture, as broad spectrum agricultural insecticides, and in timber treatment (PCP). Use of these products was restricted from the 1970s onwards and import, manufacture, and sale prohibited by 1989 (1991 for PCP) (Table A2.2). PCBs are another group of POPs – these are also organochlorines and were widely used in industrial applications due to their stability and nonflammability. PCBs were not manufactured in New Zealand but were imported, particularly in the electricity industry within transformers and capacitors, before their regulation from 1986 onwards. Dioxins and furans are a third group of POPs. While these organochlorines have never been manufactured intentionally, they occur as by-products, particularly in the manufacture of PCP, PCBs, and the pesticide 2,4,5-T (produced in Taranaki), during chlorine treatment of pulp and paper, and released during combustion, especially of waste.

While these POPs were most heavily used in the post-war period, before restrictions in the 1980s and 1990s, there are additional organic compounds that have been more recently identified as POPs, with usage then restricted. This includes some brominated flame retardants, including polybrominated diphenyl ethers (PBDEs) and brominated cyclododecanes. These compounds were not produced in New Zealand but were used in manufactured items including polyurethane and polystyrene foams, especially in furniture and upholstery. Detection of these compounds in many environmental matrices, but particularly within blood, led to their regulation worldwide (listed under the Stockholm Convention on Persistent Organic Pollutants in 2009 and 2013) and their use has been restricted in New Zealand since around 2016.

### Toxicology

POPs readily accumulate in biota, particularly in lipid-rich tissues and organs (U.S. Environmental Protection Agency 2009). DDT and other OCPs act as neurotoxins, disrupting the central nervous system, resulting in behavioural changes including spasms, lethargy, or hyperactivity (Beckvar & Lotufo 2011). This mode of action is more often associated with lethal effects. Sublethal effects of OCPs include those considered as endocrine responses, such as reduced reproduction, abnormal sex ratios in offspring, morphological abnormalities in gonads, and altered sex steroid levels (Beckvar & Lotufo 2011).

DDT and many other OCPs are highly toxic to aquatic biota, including fish. For example, LC50 concentrations (the water concentration that would be lethal to 50% of exposed organisms) for DDT for marine fish species range from 0.26 to 10 µg/L (ANZECC & ARMCANZ 2000). This would place DDT in the most toxic category based on the New Zealand EPA’s classification (Environmental Protection Authority 2021). The tissue residue concentrations associated with adverse effects are mainly in the range 1–10 mg/kg wet weight (Beckvar et al. 2005).

These concentrations are well above any that would be expected to be measured within coastal waters in New Zealand, including the Hauraki Gulf, based on the monitoring of river water and freshwater fish species (see Ministry for the Environment 1998 and below). The lack of detection of DDT and other OCPs in livers tested from affected snapper suggests that there is low likelihood of the adverse effects being due to DDT, assuming snapper do not have extremely low sensitivity.

**Table A2.2: History of POP usage and regulation in New Zealand (summarised from Buckland et al. 1998).**

| Pollutant | Main uses | Regulation |
|---|---|---|
| PCBs | Industrial applications, predominantly in electrical industry | Import was prohibited from 1986, use and storage prohibited in 1995 |
| DDT | Pasture insecticide, widely used in agriculture, market gardens, parks, and lawns from 1951 onwards. Heavily used in New Zealand | Application type controlled in 1961, pasture use banned in 1970, deregistered in 1989 |
| Dieldrin and aldrin | Used in sheep dip & sprays, also for grass grub control and in timber treatment. Heavily used in New Zealand | Regulated from 1961 for stock treatments, other applications regulated 1964–1970s, deregistered in 1989 |
| Chlordane | Largely used in timber industry | Deregistered in 1989 for sale, manufacture, or use |
| Hexachlorobenzene | Seed dressing fungicide | Deregistered in 1972 for sale, manufacture, or import |
| Heptachlor, endrin, toxaphene | Insecticides but not widely used in New Zealand | Deregistered in 1989 for sale, manufacture, or use |
| Lindane | Insecticide for lice control on cattle, and for household uses | Controls (uses, types, records) added from 1961, but only deregistered in 1990 for sale, manufacture, or use |
| PCP | Timber treatment (antisapstain) | Use in timber treatment industry voluntarily ceased in 1988, deregistered as pesticide in 1991 |
| Dioxins and furans | Unintentionally produced during pesticide manufacture, released during pulp and paper manufacture with chlorine bleach, and in combustion, especially waste incineration | Controls in use of 2,4,5-T and PCP from late 1980s, changes in waste incineration and pulp and paper processes from 1990s |

## Occurrence

Organochlorines were regularly monitored by several regional councils during the 1990s and 2000s, particularly in sediment and biota (where they tend to accumulate) and were also the focus of a national sampling programme to establish background concentrations in air, soils, rivers, and estuaries (Ministry for the Environment 1998). The latter showed these contaminants were present at low concentrations throughout New Zealand, with some locations having higher concentrations (Ministry for the Environment 1998). DDT was detected in eels and trout from all locations, with lowest concentrations in the Haast River and highest concentrations in the Halswell River, Canterbury (0.2 mg/kg wet weight) (Ministry for the Environment 1998).

Auckland Council (or more accurately, its predecessor Auckland Regional Council) measured OCPs in estuarine sediments throughout the region in 2003, 2007, and within rural estuaries in 2009/2010. DDT (total DDT) and dieldrin were detected at 12 of the 27 sites in 2003, with concentrations highly variable between sites (Mills 2014). Endosulfan (and a key degradation product), hexachlorobenzene, lindane, and aldrin have also been detected at times at some sites (Mills 2014). Repeated sampling at several sites in 2007 showed lower concentrations compared with 2003, suggesting OCP concentrations were likely to be variable at any location (Mills 2014). Sampling in rural harbours and estuaries showed low concentrations of OCPs and PCBs, close to the concentrations found in laboratory blanks (Mills 2014). The estuaries sampled included Whangateau and Mahurangi harbours and the Whitford Embayment – locations with rural and forest landuse in the catchment where OCPs would likely have been used (Balneaves 1981, Gous 2003). There has been insufficient sampling at the same locations, using the same methods, to allow any assessment of trends over time for these contaminants (Mills 2014).

Auckland Council (and previously Auckland Regional Council) also ran a Shellfish Contaminant Monitoring Programme (SCMP) from 1987 to 2013. This programme measured OCPs (pesticides and PCBs) in oysters in the Manukau Harbour and mussels in the Waitemata and Manukau harbours and Tamaki Estuary from 1999–2000. A review of this programme in 2013 showed OCP concentrations had generally decreased over time, with the exception of DDT in mussels for which there were no significant trends over time (Figure A2.2) (Stewart et al. 2013). The general decrease in OCPs reflects trends seen internationally, for example in the US Mussel Watch Programme (Kimbrough et al. 2008) and the French Réseau National d'Observation (RNO) de la Qualité du Milieu Marin data (RNO 2000, 2006).

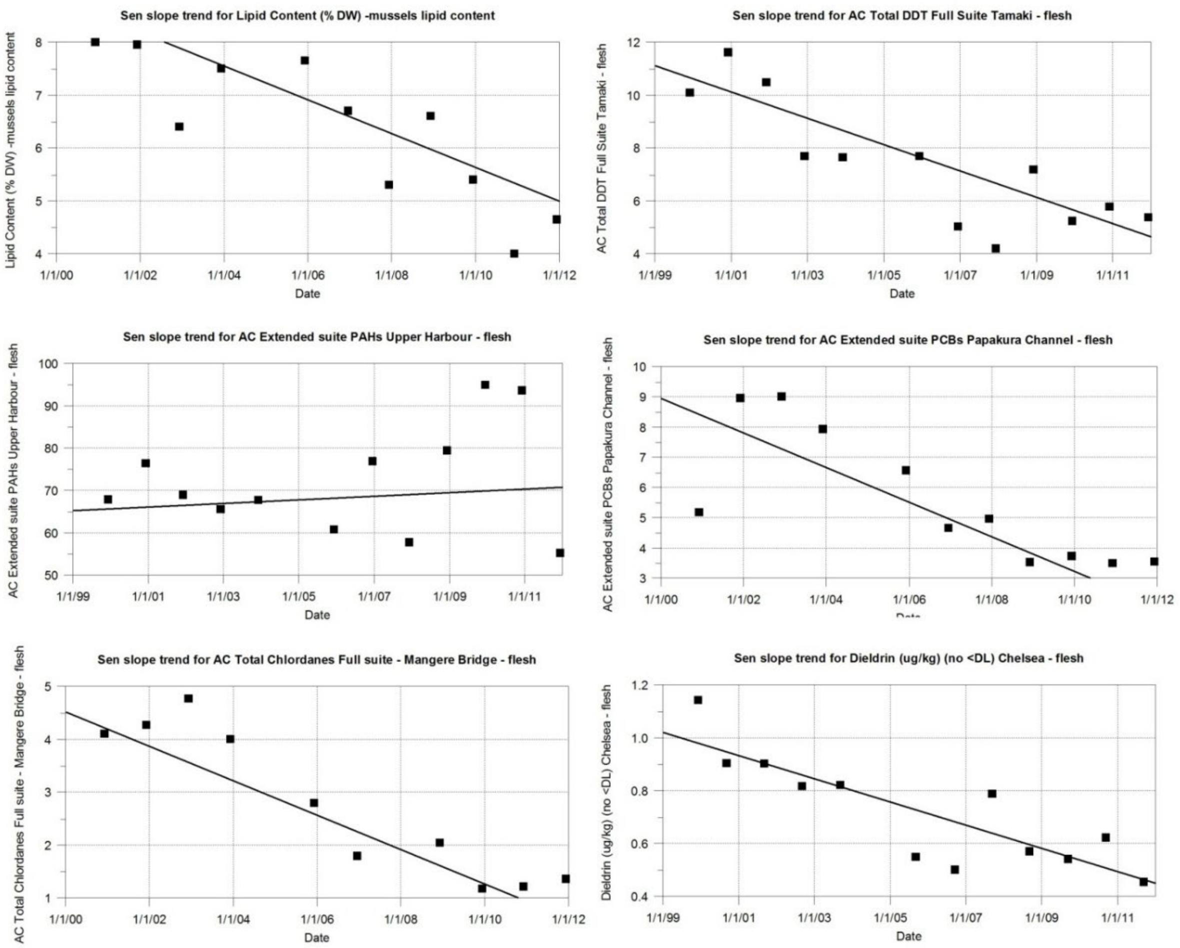



**Figure A2.2: Trends in lipid content and residues of some organochlorine pesticides in mussels deployed in Auckland harbours (from Stewart et al. 2013). Sourced from Auckland Council, subject to copyright.**

In addition to the broader monitoring undertaken by Auckland Council, Ports of Auckland have measured DDT and other OCPs in the sediments around the Port of Auckland at regular intervals to support dredging operations. These data indicate that there are hot-spots of contamination around the wharves, with DDT the most frequently detected and exceeding guideline values at times (Golder Associates (NZ) Ltd 2018).

The concentrations of OCPs detected around the Auckland region do not appear to be significantly different to those measured at other locations around New Zealand, including Napier, Wellington, Nelson, and Christchurch, with the exception of occasional hot-spots (Roberts & Forrest 1999, Strong 2005). Greater Wellington Regional Council has measured OCPs in sediments from Porirua and Wellington harbours on several occasions and found generally low concentrations, with a few locations where guideline values were exceeded (Sorensen & Milne 2009, Milne 2010, Oliver 2014, Oliver & Conwell 2014). They reported higher DDT concentrations in intertidal sediments compared with subtidal and also noted that DDT concentrations had decreased since 2004. Surveys in the Bay of Plenty suggest POPs are at slightly lower concentrations in this region, e.g., Park (2014). Of all the POPs measured around New Zealand, DDT (or its metabolite DDE) is the most frequently detected and is typically found at the highest concentrations.

PBDEs are not regularly monitored but have been examined in sediments around Auckland (Stewart et al. 2016) where they were detected at all sites, at concentrations within the range observed internationally.

### Current-use pesticides

#### Introduction

There are a large number of pesticides that continue to be used in New Zealand. These can be grouped either by their usage: e.g., insecticides, herbicides, fungicides, and rodenticides; or by their chemical structures, such as organophosphates (all having a central phosphate molecule, examples are chlorpyrifos, diazinon, dichlorvos), pyrethroids (synthetic chemicals similar to pyrethrins from chrysanthemums, common example is fly spray), organonitrogen pesticides, carbamates, and neonicotinoids. Each of these pesticides works through different modes of action and has different target species.

Compared to metals and organochlorine pesticides, there is very limited monitoring of pesticide use, or their presence in environmental matrices (water, sediments, biota). The most recent review of pesticide use in New Zealand was in 2005 (Manktelow et al. 2005) and reported annual sales of over 300 tonnes of insecticides, around 800 tonnes of fungicides, and 2500 tonnes of herbicides. Although this review recommended a follow-up in five years' time, and also recommended that New Zealand implement a better system for recording pesticide sales and use (Manktelow et al. 2005), neither recommendation has been adopted.

A review in 2010 described only the pesticides and their uses, but not volumes (or mass) used in different sectors (Chapman 2010). Chapman (2010) identified 28 pesticide active ingredients used as insecticides on pastures and forage crops, though 10 of these were under review and some uses have been revoked (Environmental Protection Authority 2023). That total does not include the pesticides used as fungicides or herbicides, or those used in the horticulture sector.

In terms of monitoring, pesticides are monitored in groundwater every four years in national surveys conducted by ESR (e.g., Close 1996, Close & Banasiak 2023), and some regional councils have also undertaken more intensive surveys (Waikato, Canterbury). Measurements of pesticides in soils, sediments, or surface waters are very rare.

#### Toxicology

Most current use pesticides have lower toxicity to fish than organochlorine pesticides, as they have been designed to be more targeted to the insects (or fungus or plants) that they aim to control. Pyrethroids, followed by organophosphates, have the greatest acute toxicity to fish though both have low persistence and therefore are unlikely to accumulate in sediment or in biota (Table A2.3).

**Table A2.3: Summary of agricultural chemical toxicity and fate characteristics (summarised from information in Vogue et al. 1994, Solomon et al. 2013, Fulton et al. 2014, Close & Banasiak 2023).**

| Pesticide group | Mode of toxicity | LC50 values for fish for example pesticides | Persistence | Bioaccumulation | Mobility |
|---|---|---|---|---|---|
| Organophosphates | High acute toxicity, highly targeted, inhibit acetylcholinesterase resulting in neurotoxic effects | Diazinon 0.08–7.8 mg/L | Low – degrades within hours or days | Moderate to low potential, Kow values 2–4 | Low |
| Pyrethroids | Disrupt sodium in the nervous system, resulting in lethal effects | Very high toxicity to fish relative to other pesticides: e.g., Cypermethrin 0.0002–0.004 mg/L | Low – degrades within days to weeks in soil and water | Moderate to low potential, Kow values 2–4 | Extremely low to low |
| Carbamates | Inhibit acetylcholinesterase and affect nervous system | Carbaryl 0.25–20 mg/L<br>Carbofuran 0.033–0.87 mg/L | Low – degrades within days to weeks in soil and water | Low potential, log Kow values 1–3 | Low to very high, depending on compound |
| Neonicotinoids | Block acetylcholine receptors, but highly selective for insect, not fish | Imidacloprid 105–241 mg/L | Low - degrades within hours to days in water | Low potential, log Kow values <1 | Moderate |
| Organonitrogen pesticides | Depends on the individual compound | Diuron 0.5–84 mg/L | Moderately to highly persistent – degrades in months in soil and water | Moderate to low potential, Kow values 2–4 | Moderate |
| Acidic herbicides (including triazines) | Baseline narcosis | Terbuthylazine 0.8 mg/L | Moderately to highly persistent – degrades in months in soil and water | Moderate to low potential, Kow values 2–4 | Highly mobile |
| Glyphosate | Oxidative stress, immunotoxicity and neurotoxicity | 1.3 to >1000 mg/L | Low to moderate– degrades within days to weeks in soil and water | Low potential, log Kow values <1 | Low |

As current use pesticides are generally of low to moderate persistence, chronic toxicity (resulting from long-term exposure) is rare unless there is constant exposure (Fulton et al. 2014). Acute effects data are mainly related to lethal effects rather than sublethal effects, such as reduced growth and emaciation. It could be assumed that sublethal effects occur at concentrations 10–100-fold lower than the lethal effects. For most pesticides listed above, this could be in the range of 10–100 µg/L though for pyrethroids this could be as low as <1 µg/L. However, to reach concentrations of that scale in the 4000 $km^2$ Hauraki Gulf, 160 tonnes of pesticide (assuming an average depth of 40 m and a concentration of 1 µg/L) would need to be delivered to the gulf in one go. Though there is little recent information on pesticide use in New Zealand, if 300 tonnes of insecticide are used across New Zealand in a year, it seems unlikely that 160 tonnes of a single substance would enter the gulf in one short period.

#### Occurrence

As mentioned above, there are minimal data regarding the presence of pesticides in the environment. Hageman et al. (2019) reported on pesticides measured in streams in Waikato, Canterbury, Otago, and Southland during stable stream flows in summer. Sampling was conducted using both grab samples and passive samplers (Polar Organic Chemical Integrative Sampler, or POCIS, and two novel samplers). Pesticides were detected at most sites; in fact three or more pesticides were detected at more than 69% of sites (Hageman et al. 2019). Chlorpyrifos was the pesticide detected at the highest concentration in POCIS samplers and was the pesticide most frequently detected in grab samples.

Pesticides have been regularly detected in groundwater (see Close & Banasiak 2023), with water soluble herbicides often the more frequently detected. Concentrations are almost always well below guideline values for human consumption but the potential risks to other organisms are not assessed. Triazine herbicides are the most frequently detected type, which is likely attributable to their higher mobility compared with other pesticides.

Of most relevance is the work by Stewart et al. (2016) measuring pesticides in sediments around the Auckland region. The only pesticides detected were glyphosate (and in a few samples, its breakdown product AMPA) found at most locations associated with urban land uses. Sediment samples in the Bay of Plenty were also analysed for a suite of pesticides, with none detected in estuarine samples and only one pesticide detected in the river sediment sample (Crawshaw 2021). The lack of detection of pesticides in estuarine sediments is not surprising as most pesticides currently used degrade faster than the legacy pesticides and therefore do not persist and accumulate in sediments.

### Emerging organic contaminants

#### Introduction

"Emerging organic contaminants" (EOCs, or "contaminants of emerging concern") is a term used to describe a wide range of contaminants that have a variety of uses, chemical structures, properties, and toxicity. The common factor is that these contaminants have not been traditionally monitored, but advances in environmental analytical techniques has shown their presence in air, water, sediment, soils, biota, and in humans. Furthermore, there are concerns about their potential effects on ecological or human health. EOCs include per- and poly-fluoroalkyl substances (PFAS) and many other industrial compounds, plasticisers (used in plastics, but also in paints and rubbers), pharmaceuticals, and personal care products (e.g., ingredients in shampoo and soaps).

#### Toxicology

Few of the compounds considered EOCs have high acute toxicity (such toxicity would likely have restricted use); the potential risks are associated with long-term sublethal effects which were not always assessed prior to approving their use. More recent advancements in laboratory tests, including *in vitro* (within cell) bioassays may have demonstrated potential for effects on reproductive or immune systems, or at early stages of development. However, in most cases there is considerable uncertainty about the possible risks of EOCs.

### Occurrence

EOCs, by definition, have not been regularly or widely monitored, though there are an increasing number of studies that have assessed these in the environment within the last 10–20 years. Most of the studies have targeted specific regions, such as Auckland, Bay of Plenty, or Wellington, with national surveys only for groundwater (Close & Banasiak 2023). The 2018 results (Close et al. 2021) indicated their widespread presence at low concentrations (2022 data are not yet available), supporting results of a Waikato region groundwater survey (Moreau et al. 2019).

EOCs have been measured in the marine environment, particularly within surveys of estuarine or harbour sediments. A study in Auckland (Stewart et al. 2013) found plasticisers, alkylphenols, and pharmaceuticals in most sediment samples, with a maximum of 14 different pharmaceuticals detected at a site in the Waitemata Harbour. EOCs have also been detected in sediments around Bay of Plenty (Crawshaw 2021).

A 2020 report on the risks from EOCs in New Zealand aquatic food species reviewed studies conducted in marine and freshwater environments around New Zealand up to 2017 (Cressey & Pearson 2020). There were generally limited data for the concentrations of EOCs in aquatic species.

## Causality assessment

Determining the cause of ecosystem decline or adverse effects in aquatic organisms is extremely difficult as ecosystems are complex, pollution sources can be diffuse, and cause and effect relationships are not always well-known. Causal assessment frameworks can assist by providing a logic system to help in identifying likely causes of adverse effects (see Cormier et al. 2010). The US EPA developed a Causal Analysis/Diagnosis Decision Information System (CADDIS), primarily for stream ecosystems (U.S. Environmental Protection Agency 2017), which can be adapted for marine systems (Table A2.4).

Where possible, this framework has been used to assess the possible causes of milky flesh syndrome (focusing on toxic substances only), weigh evidence for each cause, reject causes that are unlikely, and identify the most probable causes (Table A2.5).

**Table A2.4: Summary of types of evidence used to assess causality (summarised from CADDIS (U.S. Environmental Protection Agency 2017)).**

| Type of evidence | The concept |
|---|---|
| **Evidence from the specific case** | |
| Spatial/temporal co-occurrence | The biological effect must be observed where and when the cause is observed and must not be observed where and when the cause is absent. |
| Temporal sequence | The cause must precede the biological effect. |
| Evidence of exposure or biological mechanism | Measurements of biota show relevant exposure to the cause has occurred, or other biological mechanisms linking the cause and effect have occurred. |
| Causal pathway | Steps in pathways linking sources to the cause can serve as supplementary or surrogate indicators that the cause and biological effect likely co-occurred. |
| Stressor-response relationships from the field | As exposure to the cause increases, intensity or frequency of the biological effect increases; as exposure to the cause decreases, intensity or frequency of the biological effect decreases. |
| Manipulation of exposure | Field experiments or management actions that increase or decrease exposure to a cause must increase or decrease the biological effect. |
| Laboratory tests of site media | Controlled exposure in laboratory tests to causes (usually toxic substances) present in site media should induce biological effects consistent with the effects observed in the field. |
| Verified predictions | Knowledge of a cause's mode of action permits prediction and subsequent confirmation of previously unobserved effects. |
| Symptoms | Biological measurements (often at lower levels of biological organisation than the effect) can be characteristic of one or a few specific causes. |
| **Evidence from other sources** | |
| Stressor-response relationships from other field studies | At the impaired sites, the cause must be at levels sufficient to cause similar biological effects in other field studies. |
| Stressor-response relationships from laboratory studies | At the impaired sites, the cause must be at levels associated with related biological effects in laboratory studies. |
| Stressor-response relationships from ecological simulation models | At the impaired sites, the cause must be at levels associated with effects in mathematical models simulating ecological processes. |
| Mechanistically plausible cause | The relationship between the cause and biological effect must be consistent with known principles of biology, chemistry, and physics, as well as properties of the affected organisms and receiving environment. |
| Manipulation of exposure at other sites | At similarly impacted locations outside the case sites, field experiments or management actions that increase or decrease exposure to a cause must increase or decrease the biological effect. |
| Analogous stressors | Agents similar to the causal agent at the impaired site should lead to similar effects at other sites. |
| **Weighing the evidence** | |
| Consistency of evidence | Confidence in the argument for or against a candidate cause is increased when many types of evidence consistently support or weaken it. |
| Explanation of the evidence | Confidence in the argument for a candidate cause is increased when a post hoc mechanistic, conceptual, or mathematical model reasonably explains any inconsistent evidence. |

**Table A2.5: Comparison of the strength of evidence for the candidate causes. Types of evidence with no evidence for any candidate cause were excluded. Codes as follows: + supports the candidate cause, - weakens the cause (additional symbols indicate the strength of support or weakness), 0 neither weakens nor supports, R refutes the cause, ne = no evidence available, na = not applicable. (Continued on next 2 pages)**

| Type of evidence | Legacy POPs | Mercury | Other metals/ metalloids | Current use pesticides | Emerging contaminants | PFAS | Comment / rationale for scores |
|---|---|---|---|---|---|---|---|
| **Evidence from the case** | | | | | | | |
| Spatial/temporal co-occurrence | - | - | - | - | - | - | Contaminant concentrations highest in intertidal areas, not in coastal and outer Hauraki Gulf where snapper are abundant. |
| Temporal sequence | --- | + | + | + | + | + | OCPs used most in 1940s–1950s – even sprayed directly onto water, with no indication of effects at that time. Concentrations of OCPs have decreased over time. Although there is potential that resuspension of sediments containing OCPs has increased exposure, or that groundwater containing OCPs has only just reached the Hauraki Gulf, it seems unlikely that this would be a source of greater concentrations than would have occurred during the usage of OCPs.<br><br>Other contaminants are either increasing in concentration or steady and therefore have potential to precede the biological effect. |
| Evidence of exposure or biological mechanism | - | ne | ne | ne | ne | ne | No OCPs detected in milky fleshed snapper (noting that detection limits may have been too high, with unknown concentrations for sublethal effects in snapper).<br><br>Other contaminants have not been tested, so there is no evidence available. |
| Causal pathway | - | ne | ne | - | + | + | Causal pathway requires contaminants to be delivered to the gulf, either in increasing concentration (for contaminants that accumulate) or as a sudden pulse. Missing steps in pathway whereby legacy contaminants contained in sediment suddenly increase in Hauraki Gulf. Missing steps for pathway where pesticides suddenly increase in Hauraki Gulf.<br><br>No evidence for increased contaminant concentrations on land or in rivers or in Hauraki Gulf during last 3 years. |

| Type of evidence | Legacy POPs | Mercury | Other metals/ metalloids | Current use pesticides | Emerging contaminants | PFAS | Comment / rationale for scores |
|---|---|---|---|---|---|---|---|
| Stressor-response relationships from the field | - | - | - | - | - | - | Similar contaminant concentrations found in Wellington/Porirua Harbour but no reports of milky fleshed snapper in this region. Weakens evidence for toxic substances. |
| Manipulation of exposure | ne | ne | ne | ne | ne | ne | No evidence available. |
| Laboratory tests of site media | ne | ne | ne | ne | ne | ne | No evidence available. |
| Verified predictions | ne | ne | ne | ne | ne | ne | No evidence available. |
| Symptoms | - | + | - | - | 0 | - | Biological response (emaciation) is broad and no other developmental or pathological effects have been noted in affected fish (e.g., deformities, lesions). Many toxicants have specific effects e.g., on development or reproduction, and lack of evidence of these characteristics in the snapper weakens evidence for those toxicants.<br><br>Mercury has been associated with emaciated fish, supporting evidence for this cause.<br><br>For emerging contaminants, organism level effects are not well-known, so the symptoms neither support or weaken evidence for this cause. |

| Type of evidence | Legacy POPs | Mercury | Other metals/ metalloids | Current use pesticides | Emerging contaminants | PFAS | Comment / rationale for scores |
|---|---|---|---|---|---|---|---|
| **Evidence from other sources** | | | | | | | |
| Stressor-response relationships from other field studies | ne | ne | ne | ne | ne | ne | No evidence as there are no other field studies that report similar biological effects to milky flesh syndrome. |
| Stressor-response relationships from laboratory studies | ne | ne | ne | ne | ne | ne | No evidence as there are no laboratory studies that report similar biological effects to milky flesh syndrome. |
| Stressor-response relationships from ecological simulation models | ne | ne | ne | ne | ne | ne | No evidence as there are no models that report similar biological effects to milky flesh syndrome. |
| Mechanistically plausible cause | - | 0 | - | - | 0 | - | Weakening evidence for OCPs as most cause developmental effects. Weakening evidence for current use pesticides as most cause nervous system effects. Weakening evidence for PFAS as deformities usually observed. |
| Manipulation of exposure at other sites | ne | ne | ne | ne | ne | ne | No evidence as there are no experiments or management actions outside of the Hauraki Gulf that report similar biological effects to milky flesh syndrome. |
| Analogous stressors | ne | ne | ne | ne | ne | ne | No evidence as there are no similar effects reported at other sites. |
| Weighing the evidence | | | | | | | |
| Consistency of evidence | --- | - | - | - | - | - | Available evidence weakens case for legacy POPs; for all other toxic substances some types of evidence support the case and some weaken. |
| Explanation of the evidence | 0 | 0 | 0 | 0 | 0 | 0 | There are no explanations for the inconsistences or ambiguities in the evidence. |

## Recommendations

At present there is limited evidence that toxic substances could be the cause of the milky flesh syndrome observed in SNA 1. On the other hand, for the most part, there is currently insufficient evidence to either support or weaken toxic substances as a cause.

Some additional studies could be undertaken that would provide evidence for or against toxic substances as likely causes. These include:

1. Measurement of toxic substances in water of the Hauraki Gulf, East Northland, and reference locations.
2. Measurement of toxic substances in fish with and without milky flesh syndrome, collected in the Hauraki Gulf and East Northland (and potentially in reference locations).
3. Measurement of biomarkers within fish with and without milky flesh syndrome.
4. Manipulative experiments with snapper exposed to toxic substances to attempt to demonstrate the syndrome under controlled conditions.

These options are described further below, with advantages and disadvantages of each approach.

### Measurement of toxic substances in water

The nature of the effects observed (milky fish being caught over an extended period of time) suggests this is unlikely to be due to an acute effect and it is more likely attributable to a contaminant occurring at a low concentration that has been present, and potentially increasing in concentration over a long time, or a non-contaminant cause. The likelihood of it being due to a one-off acute exposure to toxic substances that caused the syndrome in a large number of fish, who have survived but not recovered, is very low due to the spatial scale that affected fish cover.

There are hundreds of contaminants that may contribute toxicity to fish and that may be present in the Hauraki Gulf and in coastal waters of SNA 1 at low concentrations – either within the water column, within sediment, and/or within the biota that snapper feed on. It is not feasible to measure all potential contaminants; chemical analysis methods are tailored to the contaminants of interest, and sampling methods for a contaminant group like metals are not appropriate for organic contaminants. Furthermore, contaminants are likely to be at very low concentrations in the waters of the Hauraki Gulf, which hinders measurement.

Passive samplers can be used in marine waters to accumulate contaminants to concentrations that are more easily measured. The samplers can be deployed from any stationary object (such as a bridge or wharf) or attached to a buoy which is anchored to the seabed. Passive samplers typically target a suite of contaminants, depending on their chemical properties (metals, semi-polar organic compounds, non-polar organic compounds), and can then be analysed in a laboratory, either for contaminants of interest (e.g., see Stewart et al. 2016), or for non-targeted analysis (e.g., Hollender et al. 2023).

Passive samplers have been successfully used to target organic contaminants, including EOCs, in studies around Auckland and Bay of Plenty harbours (Stewart et al. 2016, Crawshaw 2021). Both studies detected a range of EOCs, but concentrations were lower and fewer EOCs were detected at sites further from urban land uses (e.g., within Tauranga Harbour but away from Tauranga city). It is likely that if passive samplers were deployed in the Hauraki Gulf, the concentrations of many contaminants would be below detection.

### Measurement of toxic substances in biota

A second option would be to measure toxicants within the snapper flesh or organs. This has already been undertaken for OCPs and the results suggest that OCPs are an unlikely cause of the observed adverse effects. Other contaminants that could be measured (relatively easily) include metals (especially mercury) and some EOCs (e.g., PFAS). Ideally fish should be frozen as soon as possible after collection to ensure sample integrity. If there is no relationship between the observed concentration of contaminants and the presence of milky flesh syndrome, this would weaken the evidence for that contaminant as a cause (evidence for stressor-response relationships from the field). In addition, concentrations of substances in the flesh could be compared with data from laboratory and field studies that relate tissue concentrations to adverse effects in fish (note these reported effects are unlikely to be milky flesh syndrome but should include sublethal effects). This would provide evidence for stressor-response relationships from other field studies or laboratory studies.

### Measurement of biomarkers

Another option to narrow the suite of putative toxic compounds is to assess biomarkers within the affected snapper (and trevally, and any other fish). Biomarkers are measures of responses within an organism, usually at the molecular to cellular level (e.g., steroid levels, induction of enzymes, indicators of oxidative damage; see Kroon et al. 2017 for a review of biomarkers for fish health). These are often used as "early warning" markers of effects that can occur at higher levels of biological organisation (organism, population, or ecosystem level responses). Biomarkers can be categorised as three types: biomarkers of exposure, biomarkers of effect, and biomarkers of susceptibility (Table A2.6).

**Table A2.6: Categories of biomarkers and examples.**

| Biomarker type | Description | Examples |
|---|---|---|
| Exposure | Indicates the organism has been exposed to a stressor or toxicant; ideally biomarker is specific to different types of toxicants. May not indicate adverse effect. | Metallothionein protein induction. |
| Effects | Indicates the potential for effects, usually using biochemical or physiological measurements (molecular and cellular level biomarkers) but may include morphological changes. | Hormone concentrations and ratios that indicate effects on the reproductive system.<br>Changes in gonads; spinal abnormalities. |
| Susceptibility | Ability of an organism to respond to the challenge of exposure to a specific toxic substance, including genetic factors and changes in receptors which alter the susceptibility. | Condition factor. |

In some cases, biomarkers of exposure are specific to different types of toxic substances. For example, exposure to oestrogens or oestrogen mimics may be established through measurement of sex steroid hormone ratio or vitellogenin (egg yolk precursor protein) in fish plasma. Exposure to non-polar, aromatic substances can be assessed through measuring the activation of the aryl hydrocarbon receptor (AhR) to which these substances bind before causing adverse effects.

Biomarkers of effect can also help to distinguish the cause, as these may point to different pathways of effect that have led to the emaciation observed with the milky flesh syndrome. For example, measurements of DNA damage indicate effects of mutagens, histopathological biomarkers in the liver indicate effects mediated through the liver, and cytokines can be used as pro-inflammatory biomarkers in samples such as blood.

There are many potential biomarkers that could be assessed, that may relate to one or many different toxic substances. If this is considered a potentially useful thing to do, then a first step would be to review possible biomarkers that a) would be useful and b) can be readily measured (ideally in New Zealand to reduce potential degradation of fish samples during transport).

## Manipulative experiments

Experiments such as exposing snapper to a toxic substance (within a tank) and observing the effects could be very useful to determine the cause of milky flesh syndrome. However, at present the list of potential toxic substances is too long. This option could be useful when there is more evidence for a particular substance. Such an experiment would provide very strong evidence under the evidence type "Manipulation of exposure". However, this approach has the limitation that a certain toxic substance may inhibit nutrient uptake and consequently cause fish to become malnourished, but this is not proof that it was the cause of MWF syndrome.

A starting point for manipulative experiments could be to capture skinny fish, which are likely to be milky fleshed, and feed them to assess whether their condition can be reversed. If body condition reversal is possible, it would provide an indication that the condition is related to food abundance, suggesting that toxins are less likely to play a major causal role.

## APPENDIX 3: Snapper cannibalism

**Example of snapper showing cannibalistic behaviour. Two snapper caught on the same hook of a longline due to a snapper consuming the stomach region (containing the hook) of another snapper.**