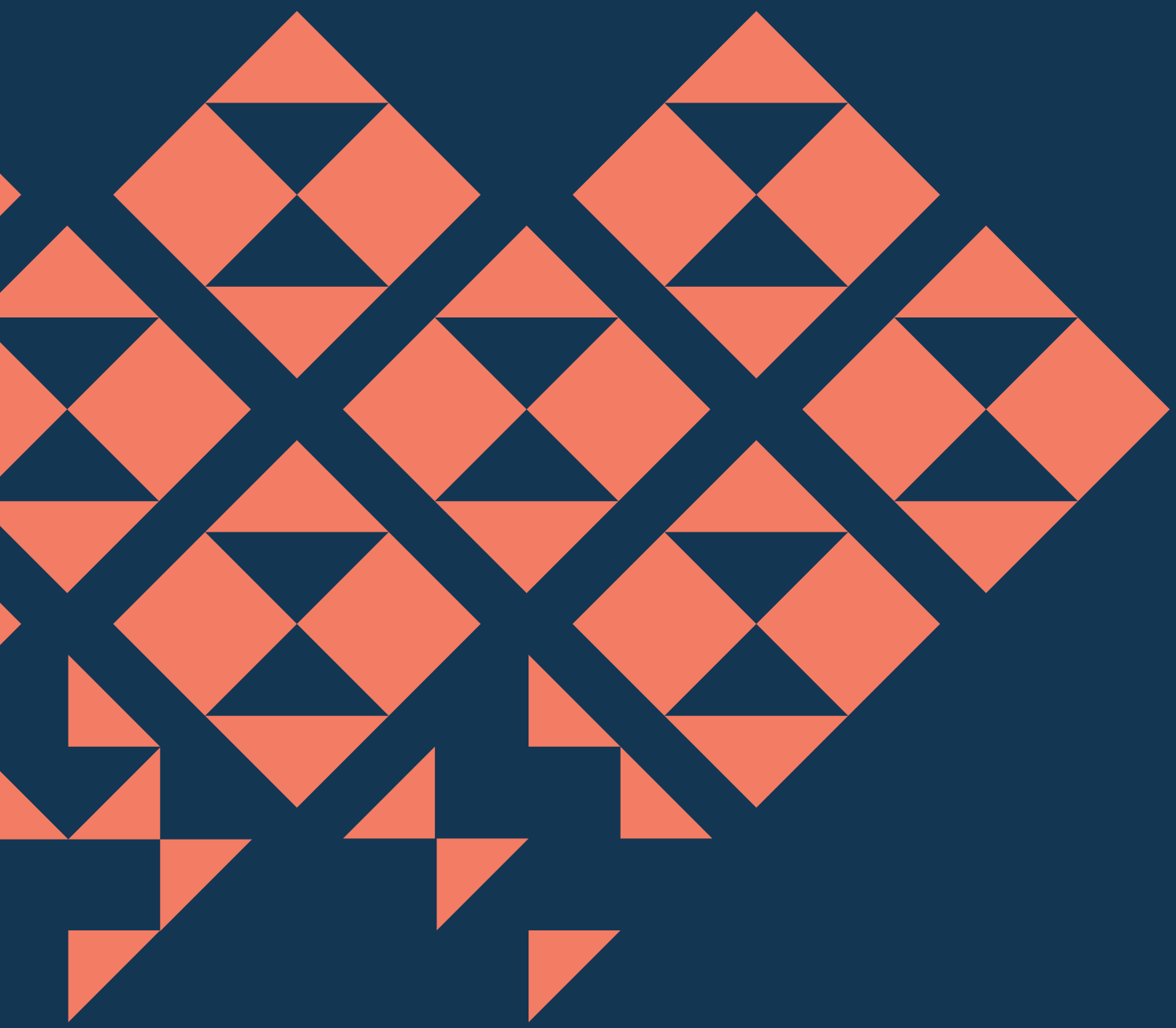


Climate Change Vulnerability Assessment of Selected Taonga Species

Supplementary Appendix 1: Species Profiles

MARCH 2020



NIWA
Taihoro Nukurangi



Te Wai Māori

Climate Change Vulnerability Assessment
of Selected Freshwater Taonga Species

Supplementary Appendix 1:
Species Profiles

Prepared for Te Wai Māori Trust

March 2020

Prepared by:
Eimear Egan
Erica Williams

For any information regarding this report please contact:

Eimear Egan
Freshwater Fish Ecologist
Freshwater Ecology
+64-3-343 7800
eimear.egan@niwa.co.nz

Contents

- Using this supplementary appendix 9**

- 1 Banded kōkopu (*Galaxias fasciatus*) 11**
 - 1.1 Habitats 11
 - 1.2 The fishery 11
 - 1.3 Life cycle, age and growth 13
 - 1.4 Distribution and environmental preferences 15
 - 1.5 Diet and predation 17
 - 1.6 Recruitment 17
 - 1.7 Current impacts of climate change 17
 - 1.8 Ecosystem level interactions 17
 - 1.9 Additional (multiple) stressors 18
 - 1.10 Critical data gaps and level of uncertainty 18
 - 1.11 References 18

- 2 Giant kōkopu (*Galaxias argenteus*) 21**
 - 2.1 Habitats 21
 - 2.2 The fishery 21
 - 2.3 Life cycle, age and growth 23
 - 2.4 Distribution and environmental preferences 24
 - 2.5 Diet and predation 26
 - 2.6 Recruitment 27
 - 2.7 Current impacts of climate change 28
 - 2.8 Ecosystem level interactions 28
 - 2.9 Additional (multiple) stressors 28
 - 2.10 Critical data gaps and level of uncertainty 28
 - 2.11 References 28

- 3 Īnanga (*Galaxias maculatus*) 32**
 - 3.1 Habitats 32
 - 3.2 The fishery 33
 - 3.3 Life cycle, age and growth 34
 - 3.4 Distribution and environmental preferences 39
 - 3.5 Diet and predation 41

3.6	Recruitment	42
3.7	Current impacts of climate change	43
3.8	Ecosystem level interactions	45
3.9	Additional (multiple) stressors.....	46
3.10	Critical data gaps and level of uncertainty	46
3.11	References	46
4	Kōaro (<i>Galaxias brevipinnis</i>).....	53
4.1	Habitats.....	53
4.2	The fishery	54
4.3	Life cycle, age and growth	56
4.4	Distribution and environmental preferences	58
4.5	Diet and predation.....	60
4.6	Recruitment	60
4.7	Current impacts of climate change.....	60
4.8	Ecosystem level interactions	61
4.9	Additional (multiple stressors).....	61
4.10	Critical data gaps and level of uncertainty	61
4.11	References	62
5	Piharau / Kanakana / Pouched Lamprey (<i>Geotria australis</i>).....	66
5.1	Habitats.....	66
5.2	The fishery	69
5.3	Life cycle, age and growth	69
5.4	Distribution and environmental preferences	70
5.5	Diet and predation.....	71
5.6	Recruitment	72
5.7	Current impacts of climate change.....	72
5.8	Ecosystem level interactions	72
5.9	Additional (multiple) stressors.....	72
5.10	Critical data gaps and level of uncertainty	73
5.11	References	73
6	Kātaha/Aua/Yellow-eye mullet (<i>Aldrichetta forsteri</i>)	76
6.1	Habitats.....	76
6.2	The fishery	76

6.3	Life cycle, age and growth	78
6.4	Distribution and environmental preferences	79
6.5	Diet and predation.....	80
6.6	Recruitment	80
6.7	Current impacts of climate change.....	81
6.8	Ecosystem level interactions	81
6.9	Additional (multiple) stressors.....	82
6.10	Critical data gaps and level of uncertainty	82
6.11	References	83
7	Tuna / Shortfin eel (<i>Anguilla australis</i>)	85
7.1	Habitats.....	85
7.2	The fishery	86
7.3	Life cycle, age and growth	89
7.4	Distribution	92
7.5	Environmental preferences	94
7.6	Diet and predation.....	96
7.7	Recruitment	99
7.8	Current impacts of climate change.....	100
7.9	Ecosystem level interactions	104
7.10	Additional multiple stressors	104
7.11	Critical data gaps and level of uncertainty	105
7.12	References	105
8	Tuna / Longfin eel (<i>Anguilla dieffenbachii</i>).....	112
8.1	Habitats.....	112
8.2	The fishery	113
8.3	Life cycle, age and growth	115
8.4	Distribution	118
8.5	Environmental preferences	118
8.6	Diet and predation.....	120
8.7	Recruitment	121
8.8	Current impacts of climate change.....	122
8.9	Ecosystem-level interactions	122
8.10	Additional (multiple) stressors.....	123

8.11	Critical data gaps and level of uncertainty	123
8.12	References	124
9	Kākahi / Kāeo / Freshwater mussels (<i>Echyridella menziesii</i>)	129
9.1	Habitats	129
9.2	The fishery	129
9.3	Life cycle, age and growth	129
9.4	Distribution and environmental preferences	132
9.5	Diet and predation	134
9.6	Recruitment	134
9.7	Current impacts of climate change	135
9.8	Ecosystem level interactions	135
9.9	Additional (multiple) stressors	136
9.10	Critical data gaps and level of uncertainty	136
9.11	References	138
10	Kōura / Kēwai / Freshwater crayfish (<i>Paranephrops planifrons</i>)	141
10.1	Habitats	141
10.2	The fishery	142
10.3	Life cycle, age and growth	143
10.4	Distribution and environmental preferences	145
10.5	Diet and predation	147
10.6	Recruitment	148
10.7	Current impacts of climate change	148
10.8	Ecosystem level interactions	148
10.9	Additional (multiple) stressors	148
10.10	Critical data gaps and level of uncertainty	149
10.11	References	149
	Appendix A Environmental variables associated with taonga fish species distribution.....	153

Using this supplementary appendix

This climate change vulnerability assessment study was funded by Te Wai Māori Trust and the Cultural Keystone Species Research Programme (MBIE Contract No. C01X1616).

This appendix is to be read alongside the main report:

Egan, E., Woolley, J.M., Williams, E. (2020) Climate change vulnerability assessment of selected taonga freshwater species: Technical report. *NIWA Client Report: 2020073CH*. March 2020: 86 p.

Egan et al. (2020) report is supplemented by three appendices that present the methodology and background information used to inform the Climate Change Vulnerability Assessments (CCVAs). These appendices are:

Appendix 1: Species profiles – Provides a profile for each of the freshwater taonga species assessed. This information, alongside Appendix 2, was provided to each assessor to inform their expert judgement and scoring of each sensitivity attribute/species in the CCVA;

Appendix 2: Sensitivity attributes – Details the definitions, justifications, links to climate change references and scoring bins for each of the 12 sensitivity attributes used to inform the CCVAs;

Appendix 3: Descriptions and projections of exposure variables – Details the projections for the exposure variables used in the CCVA.

Using the available literature, a species profile was created for each freshwater taonga species being assessed. The species profiles covered the following topics: (1) Habitats, (2) The fishery (where appropriate), (3) Life cycle, age and growth, (4) Distribution and environmental preferences, (5) Diet and predation; (6) Recruitment, (7) Current impacts of climate change, (8) Ecosystem level interactions, (9) Additional stressors, and (10) Critical data gaps and uncertainty.

These profiles were written as stand-alone documents if Te Wai Māori Trust wish to separate them in the future. Therefore, there may be some overlap in the content of the profiles. Species profiles were created for;

- Section 1: Banded kōkopu (*Galaxias fasciatus*);
- Section 2: Giant kōkopu (*Galaxias argenteus*);
- Section 3: Īnanga (*Galaxias maculatus*);
- Section 4: Kōaro (*Galaxias brevipinnis*);
- Section 5: Piharau/kanakana/pouched lamprey (*Geotria australis*);
- Section 6: Kanae/yellow-eye mullet (*Aldrichetta forsteri*);
- Section 7: Tuna/shortfin eel (*Anguilla australis*);
- Section 8: Tuna/longfin eels (*Anguilla dieffenbachii*);
- Section 9: Kākahi/kāeo/freshwater mussel (*Echyridella menziesii*); and
- Section 10: Kōura/kēwai/freshwater crayfish (*Paranephrops planifrons*).

The NZFFD was used to prepare distribution maps for each fish species. The NZFFD was accessed on 22 May 2017 and contained 42,133 records of fish occurrence from throughout Aotearoa-NZ (Williams et al. 2017). Each of these records contained information on sampling location, date, sampling method and the fish population. We used all the observations available (i.e., all fishing methods and dates) to display the presence/absence of each species for all the NZFFD records.

The appropriate species profile(s) was provided to an expert assessor, alongside Supplementary Appendix 2, to inform their expert judgement and scoring of each sensitivity attribute/species in the CCVA.

1 Banded kōkopu (*Galaxias fasciatus*)



Figure 1-1: Adult banded kōkopu (*Galaxias fasciatus*). Source: NIWA.

1.1 Habitats

Banded kōkopu (Figure 1-1) are found in 10 different habitat types (see Supplementary Appendix 2). As adults, their preferred habitats are pools and backwaters in slow flowing, first- to third-order tributaries and rivers, with reasonably extensive riparian vegetation providing bank cover. They are typically found at low-mid elevations in pools of small forested streams where there is good cover from overhanging vegetation, banks, boulders.

The spawning habitats of banded kōkopu comprise substrates and riparian vegetation although the suite of plant species they use is unknown. The habitats of larval and post-larval banded kōkopu are not well known. Larval banded kōkopu can develop in fresh water, in estuaries and in marine waters but their specific habitat requirements (i.e., substrate, depth) are not understood. Galaxiid larvae are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) and are most abundant up to 6 km offshore (Hickford and Schiel 2003). However, the specific distribution of banded kōkopu larvae is unknown.

1.2 The fishery

Information about the impacts of harvesting on Aotearoa-New Zealand whitebait populations is sparse and there are no indicators of the status of the fishery. The whitebait fishery is controlled by seasonal closures and gear limits, with little understanding or integration of stock structure, spatial and temporal variation in life histories or population dynamics into fisheries management (McDowall 1991; Egan et al. 2019). Seasonal closures serve to protect four migratory galaxiid species including banded kōkopu, which can make up 10-20% of the whitebait catch in some regions (Figure 1-2 to Figure 1-3) (Yungnickel 2017).

Since the 1920's, fishing pressure on whitebait has increased, meaning individual catches have declined and so there is a perception that the fishery is in decline (McDowall 1968). However, this heavy, long-term depletion of stocks has compromised the integrity of the whitebait fishery such that up to 1968, few rivers satisfied the local demands and requirements of whitebaiters (McDowall 1968). There have been few management interventions to limit declining populations. This is largely because the exact cause of declining populations is not known, and a lack of knowledge of the species biology and ecology prevents appropriate management actions being taken.

The phenotypic traits of banded kōkopu post-larvae (i.e., size, age) indicate there are several stocks in Aotearoa-New Zealand that are not accounted for in current fisheries management practices (McDowall and Eldon 1980; McDowall et al. 1994; Rowe and Kelly 2009; Yungnickel 2017).

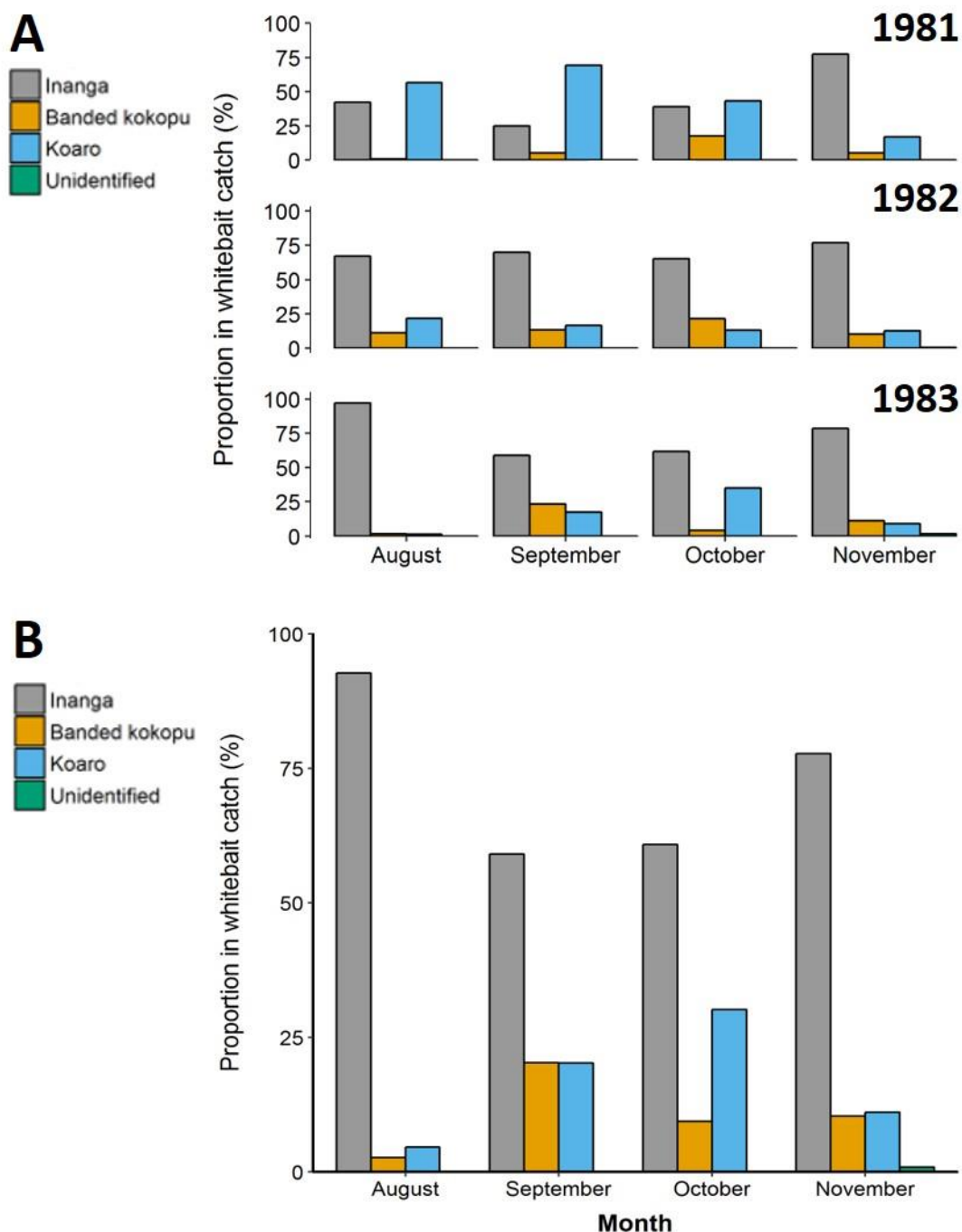


Figure 1-2: (A) Monthly variation in whitebait species composition across 12 Bay of Plenty rivers between 1981–1983; and (B) Average monthly variation in species composition between 1981–1983 for 12 Bay of Plenty rivers. Source: Redrawn by Egan from data in Rowe et al. (1992).

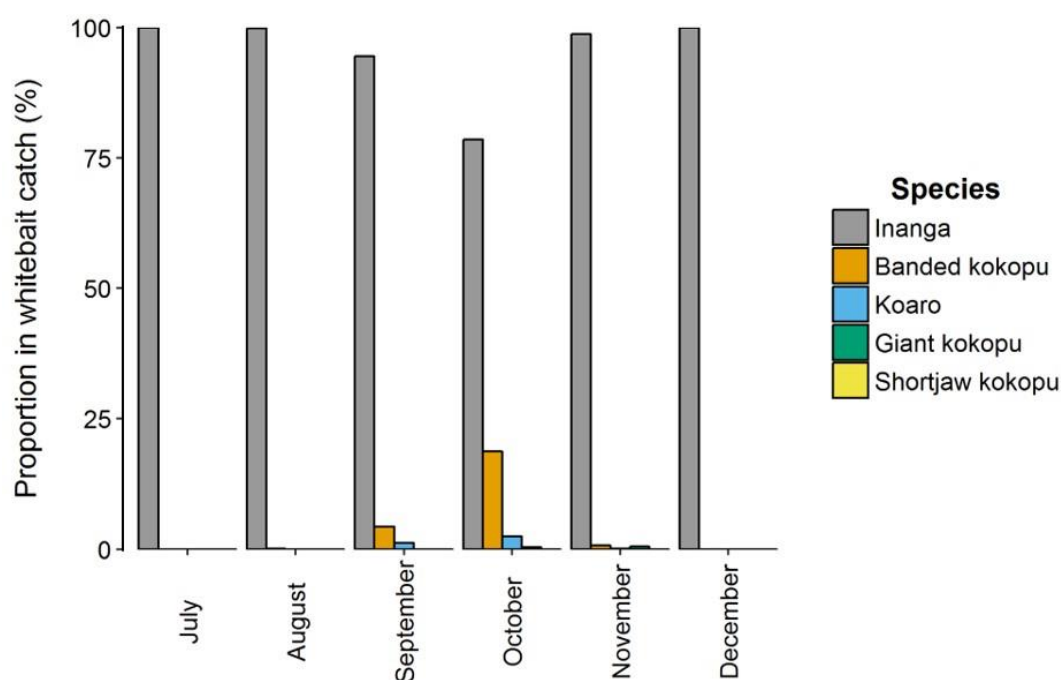


Figure 1-3: Proportion of banded kōkopu in the whitebait catch from the Waikato Region for 2015.
 Source: Data from Yungnickel (2017) redrawn by Egan (2017).

1.3 Life cycle, age and growth

Banded kōkopu is a large, amphidromous galaxiid (with females reaching up to 300 mm in length), which also readily forms land-locked populations in lakes. Specimens are commonly found up to 200 mm (McDowall 1990), which are likely to be about 7 years of age based on average growth rates (Hopkins 1979). This is likely to be the typical intergeneration time of this species, assuming this is the common adult length and growth rate. Banded kōkopu can often reach up to 260 mm (McDowall 1990), which are likely to be around 10 years old. Individuals are however likely to reach older ages given that West et al. (2005) found one specimen that may have been up to 15 years of age. This is the oldest known specimen and is likely to be close to the maximum lifespan of this species.

Spawning occurs from autumn/early winter close to typical adult habitats. Eggs have been observed to be deposited terrestrially during high water events, hatching after re-inundation (Mitchell and Penlington 1982). Recent observations of banded kōkopu spawning were made on riparian vegetation in an agricultural stream following a large flood event (Figure 1-5). Larvae rear in estuaries or at sea and return to fresh water after 4–6 months as whitebait. This species is a skilled climber at the whitebait stage and is able to migrate up substantial waterfalls (Mitchell and Penlington 1982).

Most life history information for banded kōkopu comes from Hopkins' (1979) study in Banks Peninsula and the Wellington Region. In this study, adults were aged and von Bertalanffy's growth equations calculated using age estimates derived from otoliths and fish lengths. Growth rates of males and females differed between regions which was attributed to differences in habitat quality (Hopkins 1979).

The first observation of banded kōkopu spawning was in 1975 by Ots and Eldon (1975). They identified downstream migrating larvae in June from a tributary of Lake Wairarapa and also observed newly hatched larvae moving downstream in late June in Banks Peninsula. Mitchell (1994) located the

spawning sites of banded kōkopu at Waahi beach reservoir. The key life history attributes of banded kōkopu are presented in Table 1-1.



Figure 1-4: Banded kōkopu riparian spawning habitats in an agricultural stream in the Waikato Region.
Source: NIWA.

West et al. (2005) tagged and recaptured banded kōkopu from several streams in Coromandel. They showed that annual growth rates of banded kōkopu ranged from 3 mm to 16.5 mm, with smaller fish growing faster than larger fish. Some individuals were slow growing as one recaptured fish grew from 195 mm to 215 mm in 7 years. They speculated that banded kōkopu greater than 200 mm were 12–15 years old. Although ageing has not been validated, these estimates are thought to be close to the maximum lifespan of this species. Growth rates differed seasonally with maximum growth occurring in summer. Growth rates also differed significantly among streams which is likely due to differences in habitat quality, but this was not explicitly tested (West et al. 2005).

Variation in body pattern and visual inspection of the gonads was used to examine the onset of sexual maturity and spawning for a population in Auckland (Hustler 2014). Variation in spawning times was found among years. In 2010, spawning occurred in May but in 2011 spawning occurred in June. Spawning fish were 100 mm to 220 mm length, but no ageing was undertaken. Banded kōkopu whitebait that enter fresh water in late winter (July/August) are usually larger and are better conditioned compared to whitebait that enter fresh water in spring (October/November). These patterns are found in both North and South Islands (Stancliff et al. 1988). Furthermore, significant differences in banded kōkopu body condition can be found among years (Stancliff et al. 1988).

Table 1-1: Key demographic characteristics for banded kōkopu.

Key demographic parameters	Estimates	Reference
Maximum size (mm)	260.0 mm	McDowall (1970)
Minimum size sexual maturity (mm)	80 mm (Male) 110 mm (Female)	Mitchell and Penlington (1982)
Age at sexual maturity (yrs)	15 yrs 7 yrs	West et al. (2005) West et al. (2014)
Longevity (yrs)	245.0 mm (Female) = 9 yrs 210.0 mm (Male) = 8 yrs	Hopkins (1979)
Growth parameter (K)	0.30 (Male) 0.48 (Female)	Hopkins (1979)
Fecundity	–	–
Larval size _{hatch} (mm)	9.1 mm 8.5 mm 9.0 mm	Ots and Eldon (1975) Charteris et al. (2003)
Egg size (mean)	2 mm 2.06 mm	Charteris et al. (2003)
Generation length (years)	–	–

1.4 Distribution and environmental preferences

Banded kōkopu is endemic to Aotearoa-New Zealand including North and South Islands, Stewart, Chatham and many offshore islands around the main islands, but not the sub-Antarctic islands (McDowall 1990). Banded kōkopu have a coastally restricted distribution, similar to īnanga, but are not as widely spread throughout the country (Figure 1-6).

Banded kōkopu present around Banks Peninsula, and they are commonly found South of Dunedin and along the West Coast of the South Island. In the North Island, banded kōkopu are commonly found north of Wellington and on the North Taranaki Coast. This species is also commonly found around Auckland and between Whakatāne and the Coromandel (Figure 1-5). Banded kōkopu can penetrate well inland to small first order streams, some of which may be above falls as high as 20 m.

Galaxiid larvae are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) but are abundant up to 6 km offshore (Hickford and Schiel 2003). No specific information on the distribution of banded kōkopu larvae exists.

Previous laboratory work has shown that banded kōkopu temperature preferences during the post-larval migration phase is centred on 16.1°C with an interquartile range of 14.8–17.7°C (Richardson et al. 1994). In an analysis of freshwater fish temperature tolerances, the most sensitive species were adult banded kōkopu (CTM = 30°C, LT₅₀ = 29.0°C, acclimated at 16°C; Olsen et al. 2012). Banded kōkopu prefer stream temperatures of 12–18°C. When dissolved oxygen levels in the water decrease, if possible, it is likely that juvenile banded kōkopu will leave the water body in search of a more ideal

habitat¹. Natural pollutants such as ammonia, which occurs in livestock waste and poorly treated wastewater, can adversely affect fish populations through displacement and, at high concentrations, death. In laboratory experiments banded kōkopu were found to be the second most sensitive species to ammonia toxicity. As with giant kōkopu, these fish have a preference for slightly acidic waters with a pH of less than 6.5. This explains why this species can be found in brown peat-stained waters. Environmental variables associated with banded kōkopu distribution includes shade, summer stream temperature and low flows (Table A-1; Leathwick et al. 2008).



Figure 1-5: Banded kōkopu distribution in Aotearoa-New Zealand. Locations of NZFFD records where banded kōkopu are present (black circles) and absent (grey circles). Source: NIWA.

¹https://niwa.co.nz/our-science/freshwater/tools/kaitiaki_tools/species/banded-kokopu

1.5 Diet and predation

Like other large-bodied galaxiids, the banded kōkopu is an opportunistic feeder, consuming an extensive range of terrestrial invertebrates and aquatic insect larvae such as caddisflies and mayflies, taken from drift or the stream bed (West et al. 2005). West et al. (2005) also found that the banded kōkopu consume native frogs (*L. hochstetteri*).

1.6 Recruitment

Little is known about the cues banded kōkopu use for inward migration to fresh waters, but migration is likely triggered by seasonal changes in water temperature and day length (Barbee et al. 2011) along with flood flows (McDowall 1995). Olfactory cues from other migratory galaxiids help post-larvae to select a river to return to (Baker and Hicks 2003). Baker and Montgomery (2001) found that banded kōkopu whitebait in tank trials were attracted to the pheromones (odours) released by adult banded kōkopu. This attraction was species-specific, as the whitebait were not attracted to the pheromones of kōaro or īnanga. A similar study by Baker and Hicks (2003) found that īnanga juveniles were attracted to the odour of adult īnanga, as well as adult banded kōkopu and kōaro; whereas kōaro juveniles were only attracted to the odour of kōaro adults. A laboratory study by Baker (2003) found that the presence of adult pheromones would cause banded kōkopu whitebait to override an avoidance response to suspended sediment. These studies indicated that juvenile banded kōkopu used pheromones as a cue to migrate upstream to find suitable adult habitat. It is presumed that upstream migration rate is influenced by water clarity and stream flows (Allibone et al. 1999) as well as temperature (Bannon and Ling 2004) as is known for īnanga.

McDowall (2010) suggested that spending less time as a dispersive larva, and/or exploiting oceanographic features (e.g., eddies, gyres), were two potential mechanisms that could minimise larval dispersal, but these mechanisms have not been tested. Variable and extreme weather conditions affecting spawning success and the associated abundance of whitebait have been recognised for decades but are not quantified.

1.7 Current impacts of climate change

Changes in the frequency, timing and magnitude of flood events that are predicted with climate change may alter the reproductive cues used by kōkopu species (Charteris et al. 2003) such as banded kōkopu. It is speculated that diadromous galaxiids may be affected by drought because spawning and larval migration to the sea occurs during periods of increased drought prevalence (i.e., in the summer and autumn; McDowall 1995). During drought conditions, low flows can result in gravel and sediment build up at river mouths and can block the inward migrations of the post-larval stages, with the effects of drought being exacerbated in regulated rivers (Mitchell and Davis Te- Maire 1994).

1.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between banded kōkopu and the wider ecosystem include:

- Kākahi (freshwater mussel; *Echyridella menziesii*) larvae are external parasites on fish during the first three weeks of life. If they do not find a fish host, they die. Kōaro are the main species cited as a host, but it is likely that all fish species including banded kōkopu can be hosts;

- As a mobile consumer with distinct ontogenetic shifts in diet and habitat use, banded kōkopu likely provide crucial links among production pathways of littoral, benthic and pelagic zones in lakes and marine environments.

1.9 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). Stressors potentially impacting banded kōkopu populations in Aotearoa-New Zealand were derived from the IUCN threat assessment² (n=5) included: ‘fishing and harvesting of aquatic resources [large scale harvest]’; ‘dams and water management/use [abstraction of surface and groundwater; large and small dams]’; and ‘residential and commercial development [housing and urban areas]’ (see Supplementary Appendix 2).

1.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- Degraded habitats and environmental conditions essential for spawning, survival and growth are significant risks to this species;
- Banded kōkopu stock structure is poorly resolved. Greater spatial coverage is needed to ascertain if more stocks exist along with a better integration of multiple techniques like genetic and otolith analyses.

1.11 References

- Allibone, R., Boubée, J.A.T., West, D. (1999) The ones that got away: determining whitebait movements and rates of escape. *Water and Atmosphere*, 7: 11–13.
- Baker, C.F., Montgomery, J.C. (2001) Species-specific attraction of migratory banded kokopu juveniles to adult pheromones. *Journal of Fish Biology*, 58: 1221–1229.
- Baker, C.F. (2003) Effect of adult pheromones on the avoidance of suspended sediment by migratory banded kokopu (*Galaxias fasciatus*) juveniles. *Journal of Fish Biology*, 62: 386–394.
- Baker, C.F., Hicks, B.J. (2003) Attraction of migratory inanga (*Galaxias maculatus*) and koaro (*Galaxias brevipinnis*) juveniles to adult galaxiid odours. *New Zealand Journal of Marine and Freshwater Research*, 37: 291–299.
- Bannon, H.J., Ling N. (2004) Running the unseen, lowland gauntlet: compounding effects of temperature, hypoxia and exercise for diadromous fishes. In: Rupp GL, White MD (eds) Proceedings of the 7th international symposium on fish physiology, toxicology, and water quality, Tallinn, Estonia, 2004. U.S. Environmental Protection Agency, Ecosystems Research Division, Athens, Georgia, USA, EPA 600/R-04/0 49: 207–218.
- Barbee, N.C., Hale, R., Morrongiello, J., Hicks, A., Semmens, D., Downes, B.J., Swearer, S.E. 2011. Large-scale variation in life history traits of the widespread diadromous fish,

² <https://www.iucnredlist.org/resources/threat-classification-scheme>

- Galaxias maculatus*, reflects geographic differences in local environmental conditions. *Marine and Freshwater Research*, 62: 790–800.
- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions* 20: 931–942.
- Charteris, S.C., Allibone, R.M., Death, R.G. (2003) Spawning site selection, egg development, and larval drift of *Galaxias postvectis* and *G. fasciatus* in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, 37: 493–505.
- Egan, E. M.C., Hickford, M.J.H., Schiel, D.R. (2019) Understanding the life histories of amphidromous fish by integrating otolith-derived growth reconstructions, post-larval migrations and reproductive traits. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29: 1391–1402.
- Hickford, M.J., Schiel, D.R. (2003) Comparative dispersal of larvae from demersal versus pelagic spawning fishes. *Marine Ecology Progress Series*, 252: 255–271.
- Hopkins, C.L. (1979) Age-related growth characteristics of *Galaxias fasciatus* (Salmoniformes: galaxiidae). *New Zealand Journal of Marine and Freshwater Research*, 13: 39–46.
- Hustler, K. (2014) Individual identification and visual indicators of breeding condition of banded kōkopu (*Galaxias fasciatus*) in an urban New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, 48: 15–23.
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- McDowall, R.M. (1968) *Galaxias maculatus* (Jenyns), the New Zealand Whitebait. *New Zealand Marine Department Fisheries Research Bulletin*, 2: 1–83.
- McDowall, R.M., Robertson, D.A., Saito, R. (1975) Occurrence of Galaxiid larvae and juveniles in the sea. *New Zealand Journal of Marine and Freshwater Research*, 9: 1–9.
- McDowall, R.M. (1990) *New Zealand freshwater fishes: a natural history and guide*. Heinemann Reed, Auckland.
- McDowall, R.M. (1991) Conservation and management of the whitebait fishery. *Department of Conservation Science and Research Series*, 38: 1–18.
- McDowall, R.M. (1995) Seasonal pulses in migrations of New Zealand diadromous fish and the potential impacts of river mouth closure. *New Zealand Journal of Marine and Freshwater Research*, 29: 517–526.
- McDowall, R.M. (2010) Why be amphidromous: expatrial dispersal and the place of source and sink population dynamics? *Reviews in Fish Biology and Fisheries*, 20: 87–100.
- Mitchell, C.P., Penlington, B.P. (1982) Spawning of *Galaxias fasciatus* Gray (Salmoniformes: galaxiidae). *New Zealand Journal of Marine and Freshwater Research*, 16: 131–133.
- Mitchell, C., Davis-Te Maire, K. (1994) Mahinga Kai Resources of the Waiau Catchment. Report to Electricity Corporation of New Zealand: 130.

- Olsen, D.A., Tremblay, L., Clapcott, J., Holmes, R. (2012) Water temperature criteria for native aquatic biota. *Auckland Council Technical Report 2012/036*.
- Ots, J.P., Eldon, G. (1975) Downstream movement of fry of *Galaxias fasciatus* Gray (note). *New Zealand Journal of Marine and Freshwater Research*, 9: 97–99.
- Richardson, J., Boubée J.A., West, D.W. (1994) Thermal tolerance and preference of some native New Zealand freshwater fish. *New Zealand Journal of Marine and Freshwater Research*, 28: 399–407.
- Rowe, D.K., Kelly, G. (2009) Duration of the oceanic phase for inanga whitebait (Galaxiidae) is inversely related to growth rate at sea. Pages 343-354. In A. Haro, K. L. Smith, R. A. Rulifson, C. M. Moffitt, R. J. Klauda, M. J. Dadswell, R. A. Cunjak, J. E. Cooper, K. L. Beal, and T. S. Avery, editors. Challenges for diadromous fishes in a dynamic global environment. American Fisheries Society, Halifax, Canada.
- Stancliff, A.G., Boubée, J.A.T., Palmer, D., Mitchell C.P. (1988) The upstream migration of whitebait species in the lower Waikato River. *New Zealand Freshwater Fisheries Report* 96: 44. NIWA, Rotorua.
- West, D.W., Jowett, I.G., Richardson, J. (2005) Growth, diet, movement, and abundance of adult banded kōkopu (*Galaxias fasciatus*) in five Coromandel, New Zealand streams. *New Zealand Journal of Marine and Freshwater Research*, 39: 915–929.
- West, D., Crow, S., David, B., Ling, N., Allibone, R., Hitchmough, R. (2014) *Galaxias fasciatus*. The IUCN Red List of Threatened Species 2014: e.T197278A2481170. <https://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T197278A2481170.en>. Downloaded on 26 March 2020.
- Yungnickel, M.R. (2017) New Zealand's whitebait fishery: spatial and temporal variation in species composition and morphology. MSc thesis, University of Canterbury, Christchurch, New Zealand.

2 Giant kōkopu (*Galaxias argenteus*)



Figure 2-1: Adult giant kōkopu (*Galaxias argenteus*). Source: NIWA.

2.1 Habitats

Giant kōkopu (Figure 2-1) are found in three different habitat types over their life cycle (see Supplementary Appendix 2). They are primarily a coastal species that does not usually penetrate very far inland. Giant kōkopu are mainly found in low altitude areas close to the south and west coasts of both main islands. Preferred habitat includes small-medium size, slow-flowing streams, although it can also occur in non-flowing aquatic habitats such as wetlands, ponds and lake margins. They are also usually found in habitats with good cover from overhanging vegetation, undercut banks, logs or debris clusters (West et al. 2014; Williams et al. 2017).

Spawning occurs adjacent to adult habitat on low-gradient banks among streamside vegetation inundated when water flows are elevated (Franklin et al. 2015). Vegetation documented as spawning habitat to date are:

- *Carex secta*;
- Wandering willie (*Tradescantia fluminensis*); and
- Yorkshire fog (*Holcus lanatus*).

2.2 The fishery

The impacts of harvesting on New Zealand whitebait populations is sparse and there are no indicators of the status of the fishery. The whitebait fishery is controlled by seasonal closures and gear limits, with little understanding or integration of stock structure, spatial and temporal variation in life histories or population dynamics into fisheries management (McDowall 1991; Egan et al. 2019). Seasonal closures serve to protect four migratory galaxiid species including giant kōkopu, however their contribution to the catch is negligible (Yungnickel 2017). The whitebait season on the West Coast was shortened in an effort to reduce the vulnerability of giant kōkopu to the fishery (West et al. 2014). New amendments to the whitebait fishery proposed by the Department of Conservation seek to

reduce the whitebait season time throughout the rest of the country with the goal of reducing harvesting pressure on giant kōkopu (Department of Conservation 2020).



Figure 2-2: Known spawning habitat of Giant kōkopu in an urban stream in Hamilton. Source: NIWA.

Giant kōkopu are considered uncommon in the whitebait catch in most parts of New Zealand and they usually run late in the season. For example, McDowall (1999) attempted to further refine the prevalence of giant kōkopu in the whitebait catches and to provide better estimates of migration timing. Giant kōkopu were only found after 4th November. Samples from only four days (all from the Hokitika River) contributed 152 of the total catch of giant kōkopu (65.8% of whitebait), and two consecutive days (11–12 December 1997) contributed 112 fish (48.5% of whitebait). In the Kaituna River in the Bay of Plenty, giant kōkopu comprised 3.7% of the galaxiid catch in November (Baker et al. 2005). In a recent assessment, the contribution of giant kōkopu to the juvenile galaxiid catch never exceeded 1% in North Island regions (Yungnickel 2017).

The whitebait fishery is currently managed as a single stock, despite these species having diverse distributions, habitat requirements, widely different life history traits (e.g., egg size, size and age at sexual maturity, fecundity, spawning times and migration patterns) and demographics. It has long been speculated that there are multiple “stocks” of whitebait in New Zealand (McDowall 1968, McDowall and Eldon 1980) and that stock identification should be a management priority to ensure the sustainability of the whitebait fishery (McDowall 1996). The existence of stocks demonstrates that local regulations are justifiable whereas the existence of one, widely dispersing stock suggests that local management is ineffective (McDowall 1991). Currently, the five galaxiid species are assumed to be one biological unit.

The stock structure of giant kōkopu remains poorly known despite this being a recommended priority for the whitebait fishery (McDowall 1996). In the Waikato River catchment, otolith analyses have confirmed that both giant kōkopu resident in lakes Waahi, Whangape and Waikare have formed lacustrine populations (David et al. 2019). This does not preclude diadromous stock from also recruiting into these catchments, however, the majority of giant kōkopu recruitment is likely to be

dependent upon larvae rearing within the lakes. In addition, otolith analyses of giant kōkopu residing in Waikato River tributaries near Hamilton indicate that these fish have reared in the Lake Waahi catchment. These recent findings indicate that the life history of the whitebait species within the Waikato River is more complex than previously thought. Although these non-diadromous populations in the upper catchment do not contribute to the whitebait fishery per se, they are of considerable conservation value because they help to sustain these species and their populations in the catchment (Egan et al. 2018; David et al. 2019).

Identification of giant kōkopu whitebait is difficult especially in parts of the country where shortjaw kōkopu and banded kōkopu co-occur. Genetic methods are needed to confirm correct identification of all whitebait species at the whitebait stage (Dijkstra and McDowall 1997) although discrimination is possible using morphological methods (Yungnickel 2017). No genetic studies have been completed for giant kōkopu, however, on-going research by Jane Goodman (University of Otago) is investigating the genetic structure of giant kōkopu and other whitebait species.

2.3 Life cycle, age and growth

Giant kōkopu spawning occurs within rivers and streams during elevated flows, usually following rainfall events (Franklin et al. 2015). Spawning has only been recorded from two sites in New Zealand, an urban stream in Hamilton and the Awaawaroa Wetland on Waiheke Island. Spawning is known to occur from late April to late June, but it possibly extends later to July and August.

Little is known about giant kōkopu spawning habits with most information to date coming from studies on a single population in the Waikato Region (Franklin et al. 2015). Currently, the known spawning vegetation is *Tradescantia fluminensis* (wandering willie), an invasive perennial herb; and *Carex secta* but it is highly likely that giant kōkopu use other species of native and exotic grasses, sedges and rushes for spawning (Franklin et al. 2015). In a landlocked population in Southland, spawning appeared to be mostly complete by the end of June, although the wide size range of juveniles caught in December suggests that spawning may take place over a considerable length of time (Rasmussen 1990). Giant kōkopu larval hatch dates were 4 July to 2 September for West Coast populations indicating spawning occurs in early June to early August (McDowall et al. 1994).

On average, giant kōkopu are 10 years at sexual maturity (B. David, WRC pers. comm.) although this has not been corroborated with otolith ageing. Franklin et al. (2015) measured egg diameter for a population in the Waikato River and found mean diameter was 2 mm but did not report how variable egg size was. Giant kōkopu fecundity estimates ranged from 8,800 to 25,700 eggs, with a mean value of 14,500 (mean fish length 28.3 mm; Jellyman 1979). Wylie et al. (2016) estimated the fecundity of giant kōkopu for aquaculture purposes. Substantial variation in egg production was found among individual fish and fecundities ranged from 2,520–15,820. On average, giant kōkopu produced 8,340 eggs with an average diameter of 1.6 mm (range = 1.5–1.8 mm).

Age and growth studies in the Hakarimata ranges and other study streams showed that most giant kōkopu were mature adults, the oldest giant kōkopu recorded was a 7+ year old female (231 mm total length; West 1989). The authors suggested that the apparent lack of juveniles was indicative of a recruitment bottleneck to the adult stages in these streams. Giant kōkopu are rarely larger than 300 mm although specimens of over 450 mm in length have been reported (McDowall 1990). The oldest recorded giant kōkopu was estimated to be 21–27 years of age from Rekohu/Chatham Island and best guesses indicate this species may live for 30 years (McDowall 1990).

Jellyman (1979) and Rasmussen (1990) studied lacustrine/lake populations of giant kōkopu in the Wellington and Southland regions. Age estimates by Jellyman (1979) found that a 336 mm length adult giant kōkopu was 12 years old while Rasmussen (1990) showed similar results and that a 323 mm male was 12 years. The oldest female found was 17 years at 382 mm and from analysis of length at age curve, fish larger than 400 mm were most likely females. Rasmussen (1990) determined that landlocked male giant kōkopu take approximately three years to reach sexual maturity while females are mature at four years. Rasmussen (1990) further estimated larval hatch dates from otoliths and identified larval hatching occurred in September. It was noted that spring spawning is commonly found among non-migratory galaxiids unlike the migratory life history types that reproduce in autumn and winter (Rasmussen 1990). Jellyman (1979) found that growth in giant kōkopu was slow after the first 2 or 3 years, and varied from 1.9 to 13.4 mm per year when fish were at lengths of between 234 and 330 mm.

Although specific data on giant kōkopu population declines are unavailable, based on historic, existing and continuing human pressures, it is likely giant kōkopu have experienced at least a 25% decline over the past 20 years (West et al. 2014). Furthermore, it is possible that large, old fecund giant kōkopu could be sustaining populations in the face of habitat loss and drain clearing mortalities and a 10- to 20-year lag may be weakening the current observations of a decline (West et al. 2014).

The key demographic characteristics of giant kōkopu are shown in Table 2-1. Giant kōkopu have flexible life histories: some individuals can rear solely in fresh waters (both lakes and rivers) while others are migratory (complete larval development in marine/estuarine environments). Otolith (fish ear bone) chemical signatures have shown that catchment-scale recruitment occurs for giant kōkopu populations in lakes (David et al. 2019). Populations from Lake Te Anau and Monowai are believed to be landlocked (Bonnett et al. 2002).

2.4 Distribution and environmental preferences

Giant kōkopu are endemic to New Zealand where it occurs in areas of suitable habitats in the North and South Islands and Chatham and Stewart Islands, but not the sub-Antarctic islands or Kermadecs (McDowall 1990). Giant kōkopu are a coastal species that appears to have a patchy distribution across Aotearoa-New Zealand over large geographic areas (Figure 2-3; Leathwick et al. 2008). In the South Island, they are predominantly absent around Fiordland and along the East Coast (with the exception a couple of small streams), apart from the South Otago Region. The most records in the South Island have come from the West Coast region. Most observations in the North Island have been recorded from around Wellington, Taranaki and the Waikato. There are also a few records from around South Auckland and Tauranga, but the remaining areas in the North Island have very few records of giant kōkopu (Williams et al. 2017).

Giant kōkopu extent of occurrence is calculated as 9,654 km² using an intersection of New Zealand Freshwater Fish Database records and spatial representations (polygons) of Freshwater Ecosystems of New Zealand 3rd–4th order river catchments and lakes (see West et al. 2014). Giant kōkopu area of occupancy is as 348 km², calculated from the sum of the lengths multiplied by estimated stream widths of river reaches in which this species was recorded the New Zealand Freshwater Fish Database (see West et al. 2014). This calculation includes lake areas where this species occurs. However, they are unlikely to use the entire lake as habitat, so the area of occupancy is likely an over estimation (D. West, pers. comm. 2014). Giant kōkopu reach their maximum levels of occurrence in catchments with a high frequency of intense rainfall events (>30 rainfall events per year; Leathwick et al. 2008). Environmental variables associated with their distribution are rain days (upstream) and flow stability (Table A-1).

Table 2-1: Key demographic characteristics of giant kōkopu. Where: * denotes estimates from non-diadromous populations and ∞ denotes studies on North Island populations

Key demographic parameters	Estimates	References
Maximum size (mm)	400–450 mm 580 mm	McDowall (1990); West et al. (2014)
Minimum size sexual maturity (mm)	–	–
Age at sexual maturity (yrs)	3 years (Male)*∞ 4 yrs (Female)* 2–3 years	Jellyman (1979); Rasmussen (1990); West et al. (2014)
Longevity (yrs)	336.0 mm = 12 yrs*∞ 323.0 mm (Male) = 12 yrs* 382.0 mm (Female) = 17 yrs*	Jellyman (1979); Rasmussen (1990)
Growth parameter (K)	–	–
Fecundity	283.0 mm = 14,500*∞ 280.0mm = 7,600* 6,514 8,340*	Jellyman (1979); Rasmussen (1990); Wylie (2011); Wylie et al. (2016)
Larval size _{hatch} (mm)	–	–
Egg size (mean)	1.74 mm*∞ 1.6 mm* 2.0 mm∞	Jellyman (1979); Wylie et al. (2016); Franklin et al. (2015)
Generation length	10 years	West et al. (2014)
Maximum age	21–27	McDowall (1990)

Galaxiid larvae are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) but are most abundant up to 6 km offshore (Hickford and Schiel 2003). No specific information on the distribution of giant kōkopu larvae exists. Trial translocations of captive reared fish to areas within the species natural range are occurring in New Zealand (Cindy Baker, pers. comm).

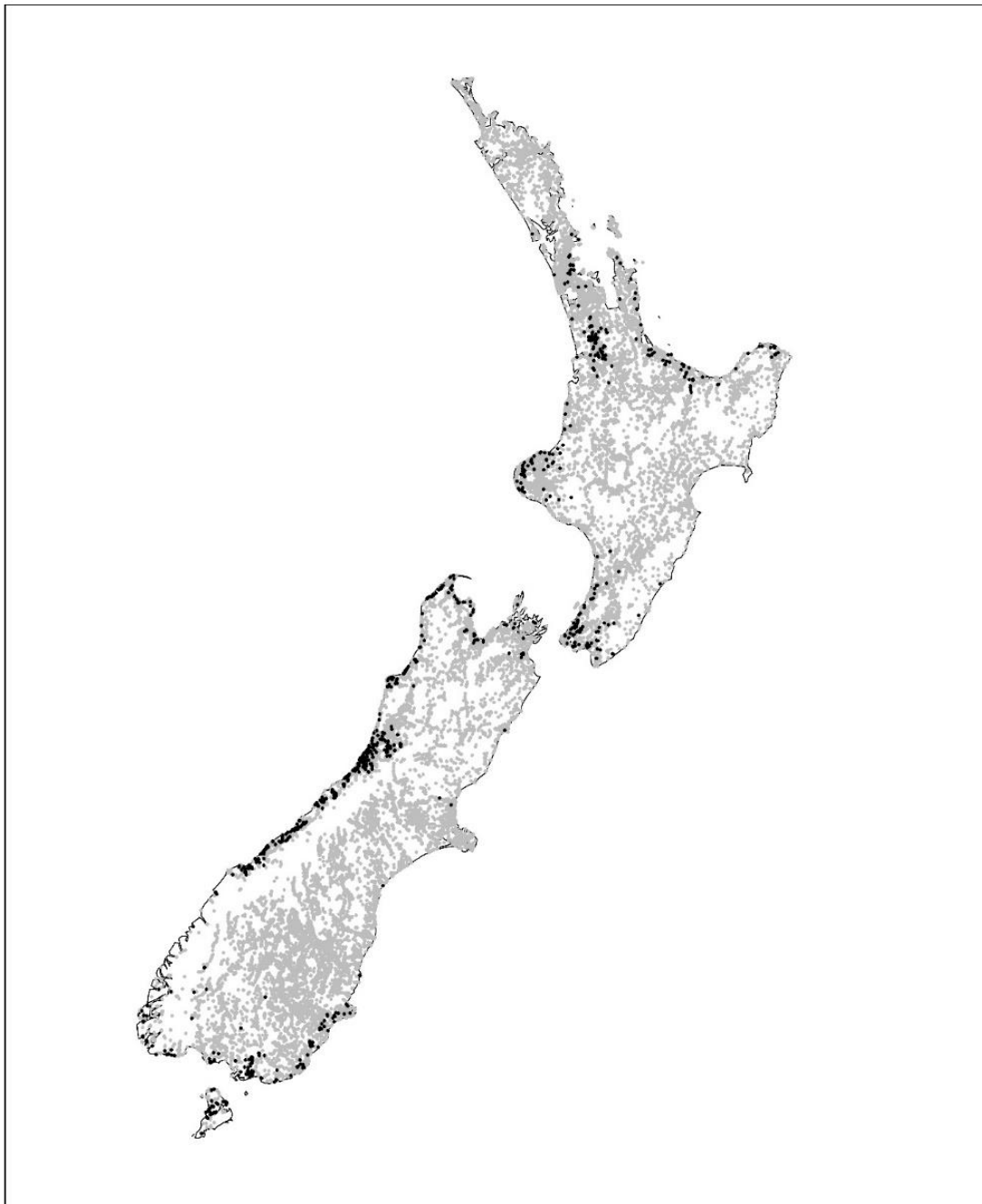


Figure 2-3: Giant kōkopu (*Galaxias argenteus*) distribution in New Zealand. Locations of NZFFD records where banded kōkopu are present (black circles) and absent (grey circles). Source: NIWA.

2.5 Diet and predation

Giant kōkopu have a diverse diet including terrestrial and aquatic invertebrates, freshwater crayfish and other fish (McDowall 1990). In one of the most detailed dietary studies, the stomach contents of 105 preserved specimens of giant kōkopu, collected from various locations around Aotearoa-New Zealand between c. 1963 and 1994, were assessed by Bonnett and Lambert (2002). Aquatic Trichoptera, Gastropoda, and Hemiptera comprised 29.1, 12.6, and 14.4% by abundance respectively, and terrestrial Coleoptera 12.2% by abundance of giant kōkopu diet. Adult terrestrial Coleoptera were

the most commonly occurring taxa and were found in 59% of the stomachs that contained food (Bonnett and Lambert 2002). Food items of aquatic origin occurred in 94.5% of the giant kōkopu stomachs that contained food and comprised 74.9% of the items by abundance.

Terrestrial food items occurred in 83.5% of the giant kōkopu stomachs that contained food and comprised 25.1% of the items by abundance. Items of terrestrial origin occurred slightly more frequently in giant kōkopu from lotic habitats than from lentic habitats (Bonnett and Lambert 2002).

Fish are a significant component of the diet, as 26% of the giant kōkopu examined contained fish remains. Giant kōkopu are probably best described as generalist feeders, as they utilise a wide range of foods of both aquatic and terrestrial origin (Bonnett and Lambert 2002).

2.6 Recruitment

Little is known about the specific cues giant kōkopu use for inward migration to fresh waters, but migration is likely triggered by seasonal changes in water temperature and day length (Barbee et al. 2011) along with flood flows (McDowall 1995). Olfactory cues from other migratory galaxiids help post-larvae to select a river to return to (Baker and Hicks 2003). Size and age of giant kōkopu post-larvae are not well described (Egan et al. 2018). In a study by McDowall (1994) giant kōkopu size and age at migration was consistent among months in several regions (Figure 2-4).

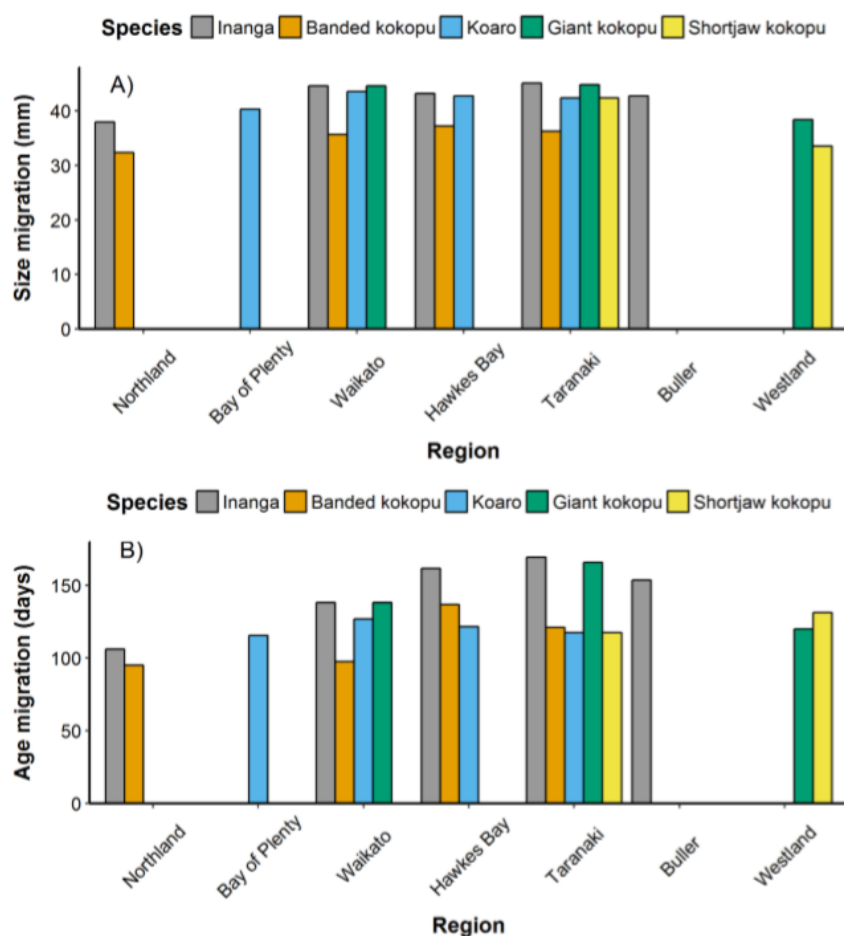


Figure 2-4: Panel (A) Mean size (mm) at inward migration of galaxiid species from several regions of Aotearoa-NZ; and Panel (B) Mean age (days) at inward migration. Source: Data reproduced by Egan from data in McDowall (1994).

2.7 Current impacts of climate change

Changes in the frequency, timing and magnitude of flood events that are predicted with climate change may alter the reproductive cues used by kōkopu species (Charteris et al. 2003) such as giant kōkopu. In years with low winter rainfall, low spawning is observed, and it is thought that the winter hydrograph triggers reproduction (Franklin et al. 2015). No other information in the literature was found.

2.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between giant kōkopu and the wider ecosystem include:

- Kākahi (freshwater mussel; *Echyridella menziesii*) larvae are external parasites on fish during the first three weeks of life. If they do not find a fish host, they die. Kōaro are the main species cited as a host, but it is likely that all fish species including giant kōkopu can be hosts;
- As a mobile consumer with distinct ontogenetic shifts in diet and habitat use, giant kōkopu likely provide crucial links among production pathways of littoral, benthic and pelagic zones in lakes and marine environments.

2.9 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). The stressors derived from the IUCN threat assessment (n=4), included ‘fishing and harvesting of aquatic resources [large scale harvest]’; ‘dams and water management/use [dams]’; ‘Invasive non-native/alien species [brown trout]’; and ‘natural system modifications’ (see Supplementary Appendix 2).

2.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- The conditions required for spawning are likely very specific which include specific hydrological cues and temperature triggers (Franklin et al. 2015) but these thresholds are not yet well quantified.

2.11 References

Baker, C.F., Boubée, J.A.T., Bowman, E., Smith, J.P. (2005) The whitebait fishery on the Kaituna River, 2004/2005. Mighty River Power.

Barbee, N.C., Hale, R., Morrongiello, J., Hicks, A., Semmens, D., Downes, B.J., Swearer, S.E. (2011) Large-scale variation in life history traits of the widespread diadromous fish, *Galaxias maculatus*, reflects geographic differences in local environmental conditions. *Marine and Freshwater Research*, 62: 790–800.

Bonnett, M.L., Lambert, P.W. (2002) Diet of giant kōkopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research*, 36: 361–369.

- Bonnett, M.L., McDowall, R.M., Sykes, J.R.E. (2002) Critical habitats for the conservation of giant kōkopu, *Galaxias argenteus* (Gmelin, 1789). *Science for Conservation*, 206: 50.
- Charteris, S.C., Allibone, R.M., Death, R.G. (2003) Spawning site selection, egg development, and larval drift of *Galaxias postvectis* and *G. fasciatus* in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, 37: 493–505.
- David, B., Chadderton, L., Closs, G.P., Barry, B., Markwitz, A. (2004) Evidence of flexible recruitment strategies in coastal populations of giant kōkopu (*Galaxias argenteus*). Department of Conservation, Wellington, New Zealand.
- David, B.O., Jarvis, M., Özkundakci, D., Collier, K.J., Hicks, A.S., Reid, M. (2019) To sea or not to sea? Multiple lines of evidence reveal the contribution of non-diadromous recruitment for supporting endemic fish populations within New Zealand's longest river. *Aquatic Conservation: Marine and Freshwater Ecosystems*. 29: 1409–1423. <https://doi.org/10.1002/aqc.3022>
- Department of Conservation (2020) Improving whitebait management. Te whakapai ake i te whakahaere inanga. *Discussion document/Puka kōrerorero*. January 2020.
- Dijkstra, L.H., McDowall, R.M. (1997) Electrophoretic identification of whitebait species. Department of Conservation, Wellington.
- Egan, E., Gee, E., Baker, C. (2018) An assessment of the Waikato River whitebait fishery with potential management options. Waikato Raupatu River Trust. *NIWA Client Report 2018036HN*.
- Egan, E.M.C., Hickford, M.J.H., Schiel, D.R. (2019) Understanding the life histories of amphidromous fish by integrating otolith-derived growth reconstructions, post-larval migrations and reproductive traits. *Aquatic Conservation: Marine and Freshwater Ecosystems* 29: 1391–1402.
- Fraser, M.M., Lalas, C. (2004) Seasonal variation in the diet of blue penguins (*Eudyptula minor*) at Oamaru, New Zealand. *Notornis*, 51: 7–15.
- Franklin, P.A., Smith, J., Baker, C.F., Bartels, B., Reeve, K. (2015) First observations on the timing and location of giant kōkopu (*Galaxias argenteus*) spawning, *New Zealand Journal of Marine and Freshwater Research*, 49: 419–426. DOI: [10.1080/00288330.2015.1045004](https://doi.org/10.1080/00288330.2015.1045004)
- Hickford, M.J., Schiel, D.R. (2003) Comparative dispersal of larvae from demersal versus pelagic spawning fishes. *Marine Ecology Progress Series*, 252: 255–271.
- Hicks, A. (2010) Native fish recruitment from riverine lakes (Chapter 10). *In*: Collier, K.J., Hamilton, D.P., Vant, W.N., Howard-Williams, C. (Eds). *The Waters of the Waikato Ecology of New Zealand's longest river*. Environment Waikato and the Centre for Biodiversity and Ecology Research (The University of Waikato), Hamilton, New Zealand: 197.
- Hicks, A.S., Jarvis, M.G., David, B.O., Waters, J.M., Norman, M.D., Closs, G.P. (2017) Lake and species-specific patterns of non-diadromous recruitment in amphidromous fish: the importance of local recruitment and habitat requirements. *Marine and Freshwater Research*, 68(12): 2315–2323.

- Jellyman, D.J. (1979) Observations on the biology of the giant kōkopu, *Galaxias argenteus* (Gmelin 1789). *Mauri Ora*, 7: 53–61.
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- McDowall, R.M. (1968) *Galaxias maculatus* (Jenyns), the New Zealand Whitebait. New Zealand Marine Department, Fisheries Research Bulletin, 2: 1–83.
- McDowall, R.M., Robertson, D.A., Saito, R. (1975) Occurrence of Galaxiid larvae and juveniles in the sea. *New Zealand Journal of Marine and Freshwater Research*, 9: 1–9.
- McDowall, R.M. (1990) New Zealand freshwater fishes: a natural history and guide. 2nd edition. Heinemann Reed, Auckland, New Zealand.
- McDowall, R.M. (1991) Conservation and management of the whitebait fishery. *Department of Conservation Science and Research Series*, 38: 1–18.
- McDowall, R.M. (1995) Seasonal pulses in migrations of New Zealand diadromous fish and the potential impacts of river mouth closure. *New Zealand Journal of Marine and Freshwater Research*, 29: 517–526.
- McDowall, R.M. (1996) Managing the New Zealand whitebait fishery: a critical review of the role and performance of the Department of Conservation. National Institute of Water and Atmospheric Research Ltd, Christchurch, New Zealand.
- McDowall, R.M. (1999) Migration season of whitebait of giant kōkopu, *Galaxias argenteus*. Department of Conservation, Wellington.
- Rasmussen, A.G. (1990) Certain aspects of the biology of a landlocked population of giant kōkopu (*Galaxias argenteus*, Gmelin 1789). MSc thesis, University of Otago, Dunedin, New Zealand.
- Rowe, D.K., Kelly, G. (2009) Duration of the oceanic phase for īnanga whitebait (Galaxiidae) is inversely related to growth rate at sea. Pages 343-354. In A. Haro, K. L. Smith, R. A. Rulifson, C. M. Moffitt, R. J. Klauda, M. J. Dadswell, R. A. Cunjak, J. E. Cooper, K. L. Beal, and T. S. Avery, editors. Challenges for diadromous fishes in a dynamic global environment. American Fisheries Society, Halifax, Canada.
- West, D.W. (1989) The ecology of native and introduced fish in some Waikato streams. Biological Sciences. MSc thesis, University of Waikato, Hamilton, New Zealand.
- West, D., David, B., Franklin, P., Ling, N., Allibone, R., Crow, S., Hitchmough, R. (2014) *Galaxias argenteus*. The IUCN Red List of Threatened Species 2014: e.T8817A3147272. <https://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T8817A3147272.en>. Downloaded on 26 March 2020.
- Williams, E., Crow, S., Murchie, A., Tipa, G., Egan, E., Kitson J., Clearwater, S., Fenwick, M. (2017) Understanding Taonga Freshwater Fish Populations in Aotearoa-New Zealand. *NIWA Client Report No. 2017326HN*. NIWA Wellington.

- Wylie, M.J. (2011) Reproductive Biology of the Giant Kōkopu *Galaxias argenteus*. MSc thesis, University of Otago, New Zealand. Retrieved from <http://hdl.handle.net/10523/2015>
- Wylie, M.J., Closs, G.P., Damsteegt, E.L., Lokman, P.M. (2016) Effects of salinity and temperature on artificial cultivation and early ontogeny of giant kōkopu, *Galaxias argenteus* (Gmelin 1789). *Aquaculture Research*, 47: 1472–1480.
- Yungnickel, M.R. (2017) New Zealand's whitebait fishery: spatial and temporal variation in species composition and morphology. MSc thesis, University of Canterbury, Christchurch, New Zealand.

3 Īnanga (*Galaxias maculatus*)



Figure 3-1: Sexually mature female Īnanga (*Galaxias maculatus*). Source: NIWA.

3.1 Habitats

Adult Īnanga (Figure 3-1) are found in a range of habitat types (n=8; see Supplementary Appendix 2). They typically prefer lowland coastal streams (Leathwick et al. 2008) and have an upper elevation limit of 230 m (David et al. 2014). Īnanga are also found in lakes and other still water body types. Īnanga were categorised as “ecological specialists” by Wedderburn et al. (2014). Ecological specialists are typically classified as a species with specific habitat requirements and/or a dependence on particular aspects of the natural flow regime (DeVictor et al. 2010).

In flowing waters, the preferred spawning habitat is dense vegetation that retains moisture (McDowall 1990; Mitchell 1990; Baker 2006; Hickford and Schiel 2011a; Hickford et al. 2017). Spawning occurs in areas that are inundated on spring tides in tidal regions of rivers, with peak spawning occurring within or near the saltwater wedge (area of saline protrusion during high tide; Goodman 2018). Plant species that are favoured for spawning were summarised by Goodman (2018) and include:

- Carex species;
- Creeping bent (*Agrostis stolonifera*);
- Mercer grass (*Paspalum distichum*);
- Kikuyu (*Pennisetum clandestinum*);
- Wandering willie (*Tradescantia fluminensis*);
- Raupō (*Typha orientalis*);
- Tall fescue (*Schenorus phoenix*);
- Wīwī (*Juncus edgariae*);
- Yorkshire fog (*Holcus lanatus*);
- Spiderwort (*Tradescantia albiflora*).

Landlocked Īnanga use the littoral (lake shore) vegetation for reproduction and larval development occurs in the limnetic zone (lake surface waters away from shore; Chapman et al. 2006). The habitats of larval and post-larval Īnanga are not well known. Larval Īnanga can develop in fresh water, in estuaries and in marine waters (see section 3.3) but their specific habitat requirements (i.e., substrate, depth) are not well understood. Galaxiid larvae are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) but are most abundant up to 6 km offshore (Hickford and Schiel 2003).

3.2 The fishery

Īnanga whitebait (Figure 3-2) support substantial commercial, recreational and customary fisheries in Aotearoa-New Zealand and to a lesser extent Australia and Chile. The impacts of harvesting on Aotearoa-New Zealand populations is sparse and there are no indicators of the status of the fishery. The whitebait fishery is controlled by seasonal closures and gear limits, with little understanding or integration of stock structure, spatial and temporal variation in life histories or population dynamics into fisheries management (McDowall 1991; Egan et al. 2019). Seasonal closures serve to protect four migratory galaxiid species (kōaro and kōkopu species; Egan et al. 2019), but not Īnanga, the most dominant species in most regions (Yungnickel 2017).



Figure 3-2: Whitebait stage of Īnanga prior to freshwater entry. Source: NIWA.

Since the 1920's, fishing pressure has increased meaning individual catches decline and so there is a perception that the fishery is in decline (McDowall 1968). However, this heavy, long-term depletion of stocks has compromised the integrity of the whitebait fishery such that up to 1968, few rivers satisfied the local demands and requirements of whitebaiters (McDowall 1968). There have been few management interventions to limit declining populations. This is largely because the exact cause of declining populations is not known, and a lack of knowledge of the species biology and ecology prevents appropriate management actions being taken (Charteris et al. 2003). Mark and recapture studies in the Mōkau River catchment estimated that between 3% and 27% of Īnanga were captured by fishers during their upriver migration (Baker and Smith 2015). Similar recapture rates were achieved in the Awakino and Oparau Rivers (Allibone et al. 1999; Baker and Boubée 2003).

The relationship between whitebait catches and numbers of whitebaiters has been used to examine the effects of varying fishing pressure on whitebait catches. However, species-specific relationships are not known. In the Rakaia and Waitaki Rivers in Canterbury, Unwin and Davis (1983) found that the total daily catch of whitebait was positively correlated with the daily count of whitebaiters. Furthermore, 40% of whitebait caught in the Rakaia River were taken over 3 days during Labour weekend (Unwin and Davis 1983). In several Bay of Plenty rivers, the daily mean number of whitebaiters was positively correlated with the daily mean total catch of whitebait (Saxton et al. 1987).

On the Awakino River, Boubée et al. (1992) documented that the numbers of whitebaiters increased on weekends and during spring tides.

Reported catches of whitebait are highly variable and reliable long-term data are extremely difficult to obtain. In the Awakino River, the highest daily individual catch was 16.8 kg for the 9 years of records, but mostly catches did not exceed 3 kg (Boubée et al. 1992). Based on data collated since 2000, Waikato River whitebaiters tend to catch more than 1 kg of bait on 50% of the days they fish, and more than 10 kg of whitebait on 9% of days fished (NIWA unpubl. data). Anecdotal evidence suggested that 2014 was an extremely productive season for whitebait in the Waikato River. Several whitebaiters reported that catches were the best in over 50 years, and an article in the Waikato Times suggested single hauls of up to 70 kg were achieved in the most lucrative whitebait season in more than 15 years.

The phenotypic traits of īnanga post-larvae (i.e., size, age, vertebral counts, growth rates and movement history) indicate there are several stocks in Aotearoa-New Zealand that are not accounted for with respect to current fisheries management practices (McDowall and Eldon 1980; McDowall et al. 1994; Rowe and Kelly 2009; Egan 2017).

Recently, significant population structuring has been found at the catchment scale using a suite of methods including otolith microchemistry, diet reconstruction using carbon and nitrogen isotopes and estimates of post-larval size and age at inward migration (Kaemingk et al. 2019). Variation in dispersal histories, feeding histories, and phenotypes are associated with the presence or absence of an estuary (embayment) that connects rivers to the sea. Rivers associated with embayments are thought to be replenished by a higher proportion of retained (or entrained) larvae, whereas rivers without an embayment are thought to be replenished by larvae that dispersed over greater distances and/or had more heterogeneous developmental histories. However, Waters et al. (2000) investigated the genetic structure of īnanga whitebait from five sites in Aotearoa-New Zealand from the Cascade River in Westland to the Bay of Islands in Northland. No genetic differences were found among these areas suggesting that īnanga larvae are widely dispersed with considerable population exchange during their marine life stage (Waters et al. 2000) and little evidence for different stocks.

3.3 Life cycle, age and growth

Īnanga are amphidromous, their life cycle typically involves downstream larval transport, dispersal and development in the marine environment followed by inward migration of post-larvae (whitebait) to fresh water where most feeding and growth occurs (McDowall 1998; Figure 3-3).

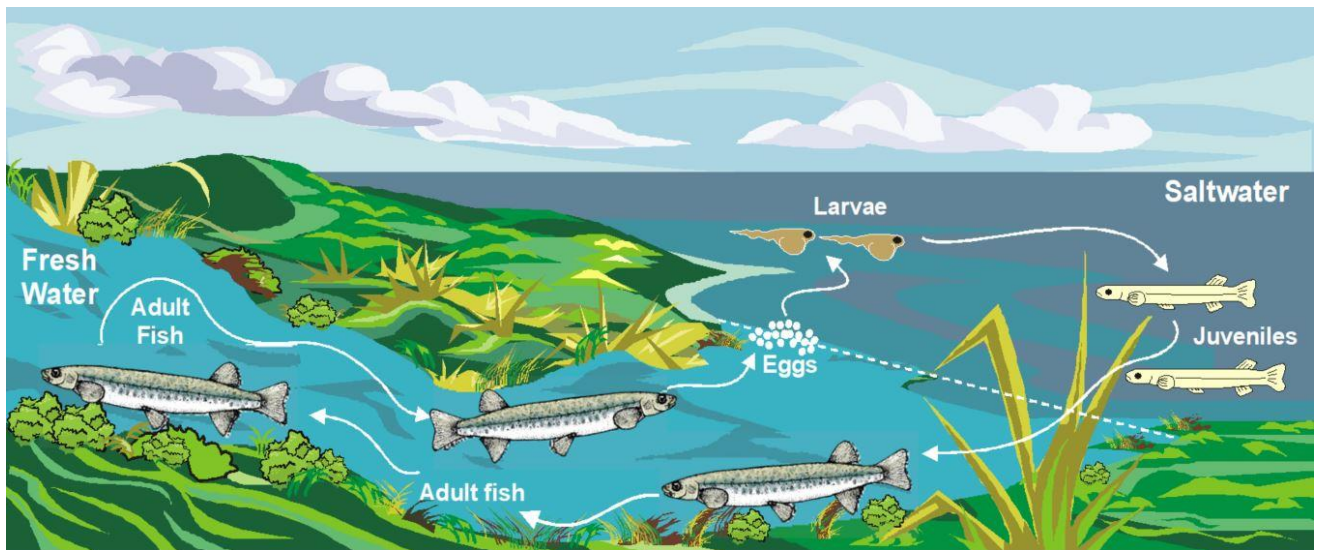


Figure 3-3: Life cycle of diadromous īnanga populations. Source: NIWA.

For diadromous populations of īnanga, mature adults (typically 50–125 mm length) move downstream to their spawning sites (McDowall 1968). Spawning occurs on riparian vegetation where the saltwater wedge penetrates fresh waters at high tides (McDowall 1988) and spawning site fidelity is observed whereby spawning occurs in the same place most years (Hickford and Schiel 2011). Spawning is linked to lunar and tidal cycles with most spawning occurring on spring-tide events. Cues like day length and seasonal changes in temperature are important for the onset of sexual maturity and spawning in īnanga (Barbee et al. 2011).

Eggs (approx. 1 mm diameter) are deposited amongst tidally inundated vegetation. Egg mortality is high, and studies show only 11% of īnanga eggs survive to hatch in compromised habitats (i.e., those damaged by stock trampling or mowing in urban areas; Hickford and Schiel 2010). Larval hatching is triggered by re-inundation of the eggs on the next tidal cycle, usually 3–4 weeks later. It is thought that larval hatching also occurs on flood flows although this has never been demonstrated (Rowe and Kelly 2009). īnanga eggs can survive for up to six weeks in the vegetation but their viability declines with longer egg development times (Benzie 1968). Newly hatched larvae, on average 7 mm length, drift downstream to the marine environment. There are few observations of īnanga in the marine environment, so little is known of their larval ecology.

īnanga whitebait migrate to fresh waters during late-winter through spring but can be observed in lower abundances throughout the year (McDowall et al. 1994). Hickford and Schiel (2016) found that less than 3% of īnanga returned to their natal stream in several rivers in the South Island. This suggests there are significant levels of mixing between river systems and less evidence for river-specific stocks. Despite this mixing, there is evidence for discrete larval pools based on regional variation in otolith-chemical signatures between the west and east coasts of the South Island. This suggests that larval dispersal between the east and west coasts of the South Island is limited (Hickford and Schiel 2016).

īnanga spend longer at sea and post-larvae migrate at a larger body size relative to the other diadromous galaxiids (McDowall et al. 1994). Average size at inward migration is 51 mm but this ranges from 36 mm to 60 mm throughout Aotearoa-New Zealand. Marine development and dispersal are poorly understood. Studies typically derive the total length of time spent in the marine environment but do not specifically derive dispersal time. Post-larvae are between 103–202 days old at migration (McDowall et al. 1994) but age at inward migration varies regionally. Rowe and Kelly (2009) found that

fish were younger in the north (Mōkau River, mean age = 135 days) compared to the south (Hokitika River, mean age = 145 days). Īnanga whitebait are on average 124 days old at inward migration to fresh water, but this can vary widely (60–187 days; Figure 3-4; Egan 2017). Growth reconstructions indicated that dispersal may be much more limited than previously assumed. For populations in some regions, the dispersal period is believed to be less than 50 days. Thereafter, an ontogenetic migration is made back to freshwater habitats (Egan 2017).

Īnanga change into the adult form in the lower reaches of rivers, while adult growth and development occurs further upstream (McDowall 1968). The rate of upstream migration to the adult habitat is influenced by stream flows among other factors such as water clarity and temperature (Allibone et al. 1999).

Īnanga are semelparous and while a few individuals can survive spawning (iteroparous), most die (Stevens et al. 2016). Size at sexual maturity, body condition and gonad weight tend to decline throughout the spawning season (McDowall 1968; Barbee et al. 2011). These patterns might be related to multiple spawning events or that the reproductive dynamics of fish that are mature later in the year differ to those that mature earlier. Generally, larger females produce more eggs but there is considerable variation in egg production among individuals. For example, up to six-fold differences in egg production were found among females that were 80 mm in length (McDowall 1968).

Recent studies from fish ear bones (otoliths; Egan 2017; McClintock 2018; Egan et al. 2019) show that Īnanga reproduction occurs from January to September and is more extensive than previously known from observations of eggs in the spawning habitats (McDowall et al. 1994a; Rowe and Kelly 2009). Significant variation in growth rates and age at sexual maturity is found among cohorts of fish born at different times of the year. Winter-hatched fish are younger at inward migration (i.e., they spend less time in the marine environment) and are faster growing in fresh water than autumn-hatched fish (Egan et al. 2019). Īnanga are an annual species with few individuals surviving to their second year (Egan et al. 2019).

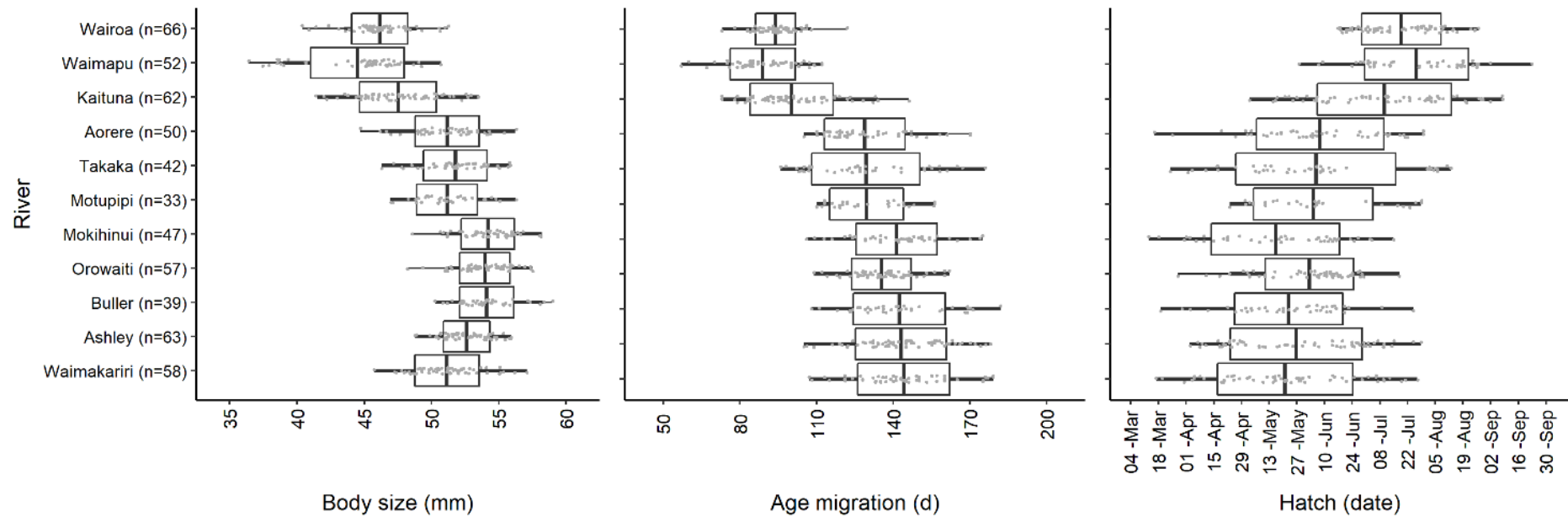


Figure 3-4: Body size (left), age at migration (middle) and hatch dates (right) for īnanga from 11 Aotearoa-New Zealand rivers. Source: Egan et al. (in prep).

Less is known about the life cycle of land-locked īnanga populations in Aotearoa-New Zealand. In Australia, land-locked populations move upstream to spawn (Pollard 1972). Spawning occurs in littoral (lake shore) vegetation and larvae rear in the limnetic zone (lake surface waters away from shore; Chapman et al. 2006). For non-migratory populations that inhabit flowing waters, elevated flows are known to trigger spawning, and this is suspected for Aotearoa-New Zealand populations also (Rowe and Kelly 2009). Upstream spawning migrations occur associated with spring flows for Australian riverine populations (Chester et al. 2014) but any association between spawning movements and the seasonal hydrograph are unknown in Aotearoa-New Zealand populations.

Considerable flexibility in the migration patterns and life histories of īnanga can be found. Freshwater larval development has been documented for īnanga from coastal populations with downstream access to the sea and upstream access to lakes (Hicks 2012). Other studies have shown that īnanga larval development occurs exclusively in the marine environment; īnanga whitebait migrate directly from the sea to fresh water, spending little time in estuaries (Hicks et al. 2005). The migratory contingent of populations are generally exposed to warmer conditions and/or less seasonal variation in temperature than the portion of the population living further inland (Leathwick et al. 2008).

Īnanga whitebait that enter fresh water in late winter (July/August) are usually larger and are better conditioned compared to whitebait that enter fresh water in spring (October/November). These patterns are found in both North and South Islands (Stancliff et al. 1988) Furthermore, significant differences in īnanga whitebait body condition can be found among years (Stancliff et al. 1988) indicative of annual variation in growing conditions during larval life. Growth during larval life is highly seasonal and this seasonality is observed in four regions of Aotearoa-New Zealand (Egan et al. in prep). The key demographic parameters for īnanga are shown in Table 3-1. Spatial demographics are not well understood; however, we have denoted values derived from North and South Island studies.

Table 3-1: Key demographic characteristics for īnanga. Where: * denotes estimates from North Island populations and + denotes estimates from South Island populations.

Key demographic parameters	Aotearoa-New Zealand estimates	Reference
Maximum body size (mm)	142.0 mm (Female)* 116.0 mm (Male)*	McDowall (1968)
Minimum size sexual maturity (mm)	50.0 mm (Female)+ 46.0 mm (Male)+	Stevens et al. (2016) Egan et al. (2019)
Age at sexual maturity (days)	264.5 days+ (81.21 mm Female) 262.8 days+ (74.86 mm Male)	Egan et al. (2019)
Longevity (yrs)	5 years	David et al. (2014)
Growth parameter (K)	NA	NA
Fecundity (range)	79-4,046 eggs+ (61.6-19.0 mm body size)	Stevens et al. (2016)
Larval size at hatch (mm)	7.1 mm ^{1*}	McDowall (1968)
Egg size (range)	0.9–1.4 mm ^{*+}	McDowall (1968); Benzie (1968)
Generation length	1 year	David et al. (2014)

3.4 Distribution and environmental preferences

Īnanga is considered the most widely distributed freshwater fish species in the world (David et al. 2014). Although Īnanga populations may be experiencing localized declines in specific areas throughout their range, this species is widespread and abundant throughout the southern hemisphere (David et al. 2014). Īnanga can be found in temperate zones (usually at low elevations up to 230 m) except for South Africa (Gomon and Bray 2011). They are found in Aotearoa-New Zealand (North and South Islands, Stewart Island and Chatham Island), south eastern and south western Australia (east and south of the Great Dividing Range from Brisbane, Queensland (28°S) to Albany, Western Australia (117°S, 50°E) and from Flinders Island and King Island, Bass Strait), Tasmania, and the southern tip of South America including Chile (from 35°S to 55°S), Patagonia, Argentina and the Falkland Islands (see David et al. 2014). Īnanga distributions are associated with stream segment summer temperatures (Table A-1; Leathwick et al. 2008).

In Aotearoa-New Zealand, this species is found close to the coast, particularly in the South Island (Figure 3-5). Larvae are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) but are most abundant up to 6 km offshore (Hickford and Schiel 2003). No specific information about Īnanga larval distribution exists. Unlike other galaxiids, Īnanga are shoaling species and highly mobile as adults.

Egg development duration decreases with increasing temperatures. For example, egg development duration decreases from 40 days to 10 days when ambient temperatures increase from 8°C to 18°C (Benzie 1968). Īnanga egg survival is unaffected by temperatures ranging from 4°C to 22°C as long as relative humidity is consistently high (see references in Hickford and Schiel (2011)). Temperature is not considered lethal to eggs unless they are fully exposed to sustained temperatures exceeding 22°C (Harzmeyer 2006). It is more likely that reduced relative humidity associated with increasing temperatures negatively affects egg development and survival (Hickford and Schiel 2011). The microclimate of the spawning habitat, which is a function of riparian species composition, affects Īnanga egg development and survival rates (Hickford and Schiel 2011). Exposure to more frequent high temperatures and subsequent reduced humidity that is associated with impacted/degraded riparian vegetation, increases egg mortality rates (Hickford and Schiel 2011). When Īnanga eggs are exposed to temperatures more than 15°C for more than 20% of the egg development time, egg survival is reduced to below 10% (Figure 3-6). High egg survival occurs mostly in undisturbed vegetation, where relative humidity is above 90%, temperature below 15°C, and where UVB radiation is filtered (Hickford and Schiel 2011).

The probability of Īnanga occurrence in fresh water is associated with temperature. Īnanga are caught most frequently at sites with warm summer temperatures ($\geq 16^\circ\text{C}$), and are usually observed where temperature seasonality is muted, i.e., maritime climates in which winters are milder than expected, given the summer temperature (Leathwick et al. 2008). Summer air temperature was the only variable significantly associated with Īnanga predicted distributions (see Supplementary Appendix 2).

Adult Īnanga can tolerate a wide range of salinities but the optimum is between 10 to 15 ppt (Ruiz-Jarabo et al. 2016). Individuals from freshwater populations (i.e., landlocked) can survive salinities between 0 to 20 ppt for at least 8 days but mortalities occur between 15 and 25 ppt (Ruiz-Jarabo et al. 2016). Individuals that are migratory can survive salinities from 5 to 25 ppt but neither landlocked nor migratory fish can survive 100% seawater (Ruiz-Jarabo et al. 2016). Artificial fertilisation experiments confirmed that successful fertilisation of Īnanga eggs only occurs at low salinities (<20 ppt; Hicks et al. 2012).

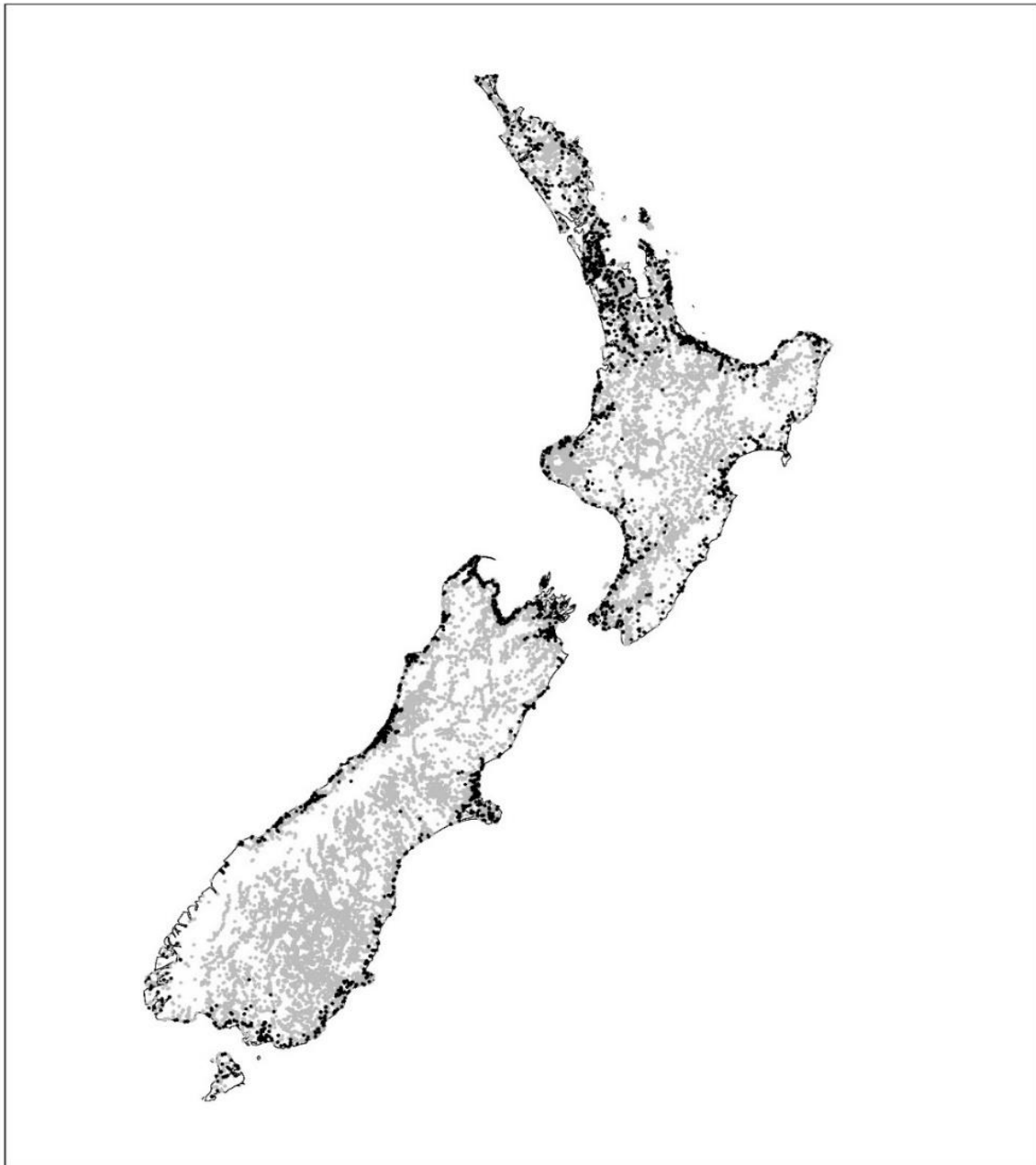


Figure 3-5: Īnanga distribution (black circles) in Aotearoa-New Zealand. Locations of NZFFD records where Īnanga are present (black circles) and absent (grey circles). Source: NIWA.

Īnanga embryos usually develop exposed to air; however, it has been shown that embryonic development can also be completed in water (Battini et al. 2000). In Australia, Īnanga can inhabit rivers with dissolved oxygen levels lower than 1 mg L^{-1} ($\sim 2.8 \text{ kPa}$; Chapman 2003) and data obtained from Aotearoa-New Zealand streams with resident Īnanga populations suggest dissolved oxygen may fall to levels as low as 3 kPa and persist for up to 6 hours (Urbina 2013). The acute lethality of low dissolved oxygen (DO) was examined in laboratory studies using several Aotearoa-New Zealand freshwater fish at 15°C (Landman et al. 2005). The 48-h LC_{50} value was used as the endpoint for acute DO sensitivity - Īnanga whitebait had LC_{50} values of $2.65 \pm 0.19 \text{ mg L}^{-1}$ (mean + SEM) and were the most sensitive species tested (Landman et al. 2005).

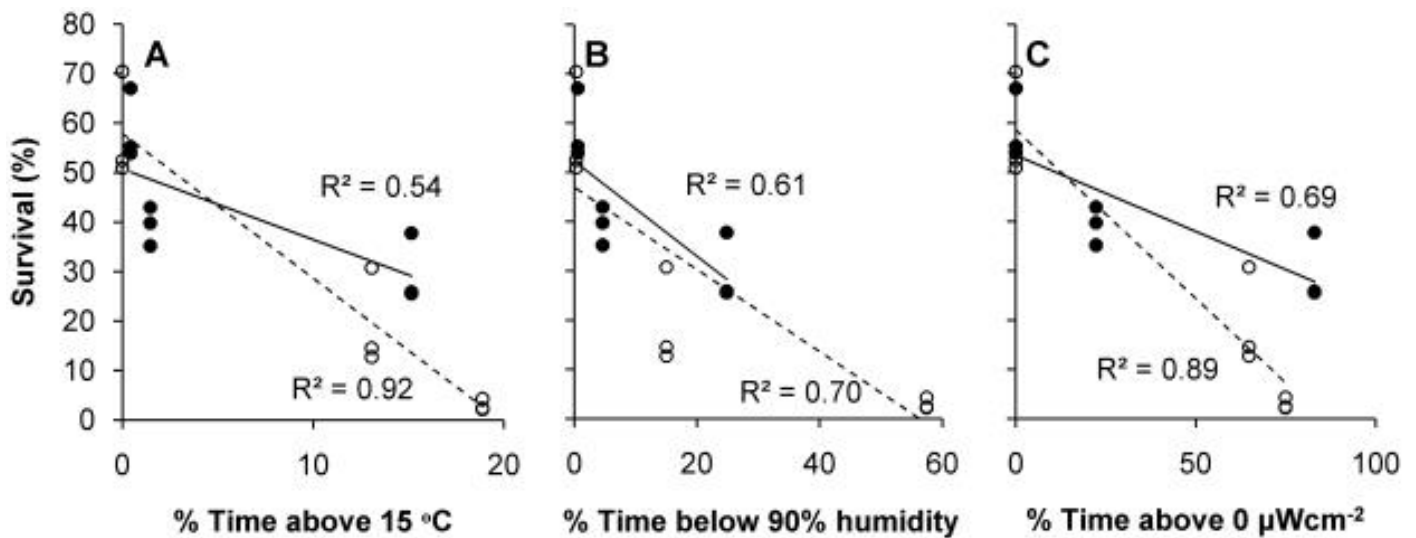


Figure 3-6: Relationship between īnanga egg survival (Y axis) and environmental variables (X axis) across all treatments. Source: Hickford and Schiel (2011). (A) time spent above 15°C, (B) time spent below 90% relative humidity and (C) time spent above 0 μWcm^{-2} UVB irradiance and survival of *Galaxias maculatus* eggs in Open (open circles) and Shaded (black circles) plots of three vegetation trimming treatments. Individual regression lines are fitted through the Open (dashed) and Shaded (solid) data points.

Īnanga cannot tolerate water with pH less than 6 and are sensitive to acidic waters (Glover et al. 2012). Jellyman and Harding (2014) did a series of 14-day experiments on five common Aotearoa-New Zealand fish species including adult īnanga to assess the effect of pH on survival and changes in body mass. No species survived in water of pH less than 4 although there was 100% survival of all adults at pH 4.5. Īnanga larvae were also tested and had high mortality at this pH. Results suggest that adults are tolerant of low-pH waters; however, the susceptibility to low pH on different life cycle stages remains unclear (Jellyman and Harding 2014).

Migrating īnanga avoid water temperatures between 29.5–31.5°C, depending on the temperature they are acclimated at (Boubée et al. 1991). Stancliff et al. (1989) reported that a temperature of 27°C was a migration barrier to īnanga because they avoided the thermal plume of Huntly Thermal Power Station by moving to the other side of the river. Bannon (2006) considered the effect of water temperature and oxygen availability on the swimming ability of three life-stages of īnanga (larvae, post-larvae and adults). Peak swimming ability was between 15–20°C for larvae (whitebait) (optimum = 17.7°C) and adults (optimum = 18.3°C), with swimming ability declining markedly at temperatures above 20°C. In contrast, maximum post-larval swimming ability occurred at 9.4°C and swimming ability declined markedly above 10°C, which was suggested to be a result from a temporary reduction in swimming ability during metamorphosis. Under mild hypoxia (75% saturation), the swimming performance of īnanga post-larvae was reduced at 15°C and 20°C, but not at 10°C when compared to normoxic conditions (96% saturation; Bannon 2006). Increases in water temperature can result in the loss of schooling behaviour in īnanga (Simons 1986), which may affect the susceptibility of īnanga post-larvae to predation.

3.5 Diet and predation

Ontogenetic differences in the diet of īnanga are known. Adult īnanga are primarily drift feeders and are commonly observed in stream sections with relatively low water velocities (0.03–0.07 ms^{-1} ; Jowett 2002). They are opportunistic feeders consuming terrestrial insects and aquatic organisms. In the Waikanae River in the North Island, McDowall (1968) found īnanga consume a high diversity of food

items; however, the three dominant aquatic food types were Copepoda (39.4%), Chironomidae larvae and pupae (21.6%) and gastropod molluscs from the Prosobranchiata order (20.7%). Adult fish also consumed larvae from the Trichoptera order (5.7%) and a range of unidentified fish eggs (2.8%). Īnanga also consume terrestrial species including Coleoptera (0.66%) and Hemiptera (0.58%). In a river population in Australia, Īnanga had a highly variable diet, no seasonality in diet was detected and Īnanga were piscivorous (Becker et al. 2008).

In the Waikato River, post-larval/juvenile Īnanga consume large quantities of Cladocera followed by Chironomidae, Saldidae and unidentified Insecta (18–24%; Catlin et al. 2019). Seasonal zooplankton production associated with spring high flow events are important for post-larval Īnanga, with cladocerans contributing 74–97% of post-larval Īnanga diet in October (Catlin et al. 2019). Large cladocerans such as *Bosmina* and *Daphnia* are preferentially selected by post-larvae as there were higher proportions in their guts relative to availability in the Waikato River (Catlin et al. 2019). Relatively little is known about Īnanga diet from lake populations. In post-glacial lakes in Argentina, Īnanga larval growth rates are related with zooplankton density and water temperature (Barriga et al. 2012). Dietary differences of post-larvae migrating into rivers with and without estuaries were inferred using stable isotopes of nitrogen and carbon values and indicated different food sources (Kaemingk et al. 2019).

In lakes, Īnanga forage in benthic and pelagic habitats and, in turn, are considered key prey species for larger piscivorous fish (Barriga et al. 2012; Cussac et al. 1992). Īnanga themselves are cannibalistic and eat their own eggs (Allibone 2003). Blue penguins (at risk declining) eat whitebait species (Fraser and Lalals 2004) but the importance of Īnanga specifically to their diet is unknown.

3.6 Recruitment

Little is known about the cues for inward migration to fresh waters, but migration is likely triggered by seasonal changes in water temperature and day length (Barbee et al. 2011) along with flood flows (McDowall 1995). Olfactory cues from other migratory galaxiids help post-larvae to select a river to return to (Baker and Hicks 2003), but most individuals do not return to their natal rivers (Hickford and Schiel 2016). Upstream migration rate is influenced by water clarity and stream flows (Allibone et al. 1999) as well as temperature (Bannon and Ling 2004).

Some studies show seasonal declines in size and age at migration, but this is not conclusive. Rowe and Kelly (2009) found no temporal differences in mean size or age in two rivers. McDowall (1968) found that seasonal patterns in size are river-dependent; the Awarua and Moeraki Rivers on the south-west coast of the South Island showed strong seasonal declines in mean size but the Taramakau River in the middle of the west coast had no seasonal patterns. Temporal declines in mean size and age have been reported in Australian (Barbee et al. 2011) and Aotearoa-New Zealand populations (McDowall et al. 1994) at the regional scale but not within a region. In a more recent study, Yungnickel (2017) mostly showed that Īnanga migrating during July were larger than those in December and January but no differences were found from September to November.

Egan et al. (2019) reconstructed the growth histories of Īnanga using the microstructure of otoliths. The authors showed that the first 50 days of marine growth were inter-dependent, indicating that early larval growth may be the critical link to understanding intra- and inter-annual recruitment variations of inward migrating post-larvae. Growth after 60 days of larval life propagated through to adult freshwater development, highlighting linkages between late marine and adult freshwater life.

McDowall (2010) suggested that spending less time as a dispersive larva, and/or exploiting oceanographic features (e.g., eddies, gyres), were two potential mechanisms that could minimise larval dispersal, but these mechanisms have not been tested.

The chemical signatures in otoliths between post-larvae recruiting to embayment (estuaries) compared to non-embayment sites indicate that īnanga are recruiting to these distinct systems 10 days before those in non-embayment sites (Kaemingk et al. 2019). Kaemingk et al. (2019) also suggest that specific phenotypic traits of marine recruits may affect the probability of survival to reproductive age in the freshwater environment. Īnanga recruiting to a river with an embayment are more phenotypically diverse compared to those recruiting to a river without an embayment. Variation in size, age, and growth rates between neighbouring rivers is thought to corresponded to different adult survivorship patterns with low survival of early hatched fish in the river with an embayment (Neilson 2016).

3.7 Current impacts of climate change

Current impacts of climate change are largely unknown for this species. Climate change and associated changes to sea surface temperature and ocean circulation patterns have been implicated in the decline of īnanga in south-west Australia (Barbee et al. 2011). However, potential changing larval dispersal patterns associated with changes in circulation are currently unknown for Aotearoa-New Zealand populations. During larval life, substantial seasonality in growth rates is observed and the lowest growth occurs in the winter months (Figure 3-7; Egan et al. [in prep]). The magnitude of this seasonal variation differs regionally with the highest summer growth rates and lowest winter growth rates found in post-larvae recruiting to the Buller region compared to other areas (Figure 3-7).

The effects of extreme and variable weather events on post-larval whitebait recruitment variation has been observed although there are no relationships that directly quantify these effects (McDowall and Eldon 1980). Droughts frequently occur on the east coast of Aotearoa-New Zealand during later summer and autumn, and their probability of occurrence is predicted to increase with climate change (MFE 2018). Drought vulnerability of 43 freshwater fish species was analysed in Australian populations using up to 14 traits for each species (e.g., fecundity, adult size, spawning temperature; Chessman 2013). Īnanga were ranked as the 16th most drought vulnerable species in the analysis.

It is speculated that diadromous galaxiids may be affected by drought because spawning and larval migration to the sea occurs during periods of increased drought prevalence (i.e., in the summer and autumn; McDowall 1995). During drought conditions, low flows can result in gravel and sediment build up at river mouths and can block the inward migrations of the post-larval stages, with the effects of drought being exacerbated in regulated rivers (Mitchell and Davis-Te Mairie 1994).

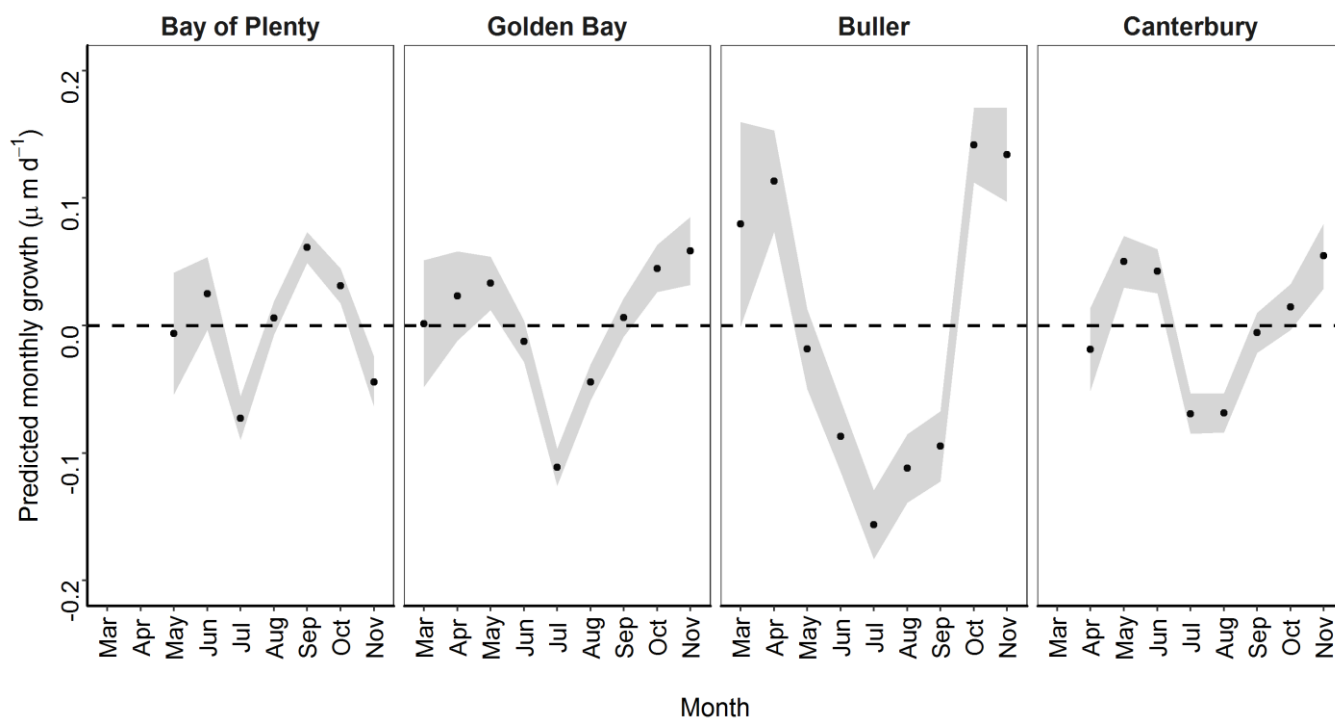


Figure 3-7: Seasonal variation in marine larval growth rates of īnanga. Source: Egan et al. (in prep).

Īnanga eggs are particularly sensitive to temperature fluctuations (Hickford and Schiel 2011) and therefore may be more susceptible to mortality under low rainfall and/or drought conditions. Extreme precipitation events and associated flooding can increase egg mortality rates because deposited sediment can suffocate the eggs (Stancliff et al. 1988a).

In Australia, īnanga were present in unregulated streams, despite prolonged drought conditions during 2006–2010 (Chester et al. 2014). Compared with historic data, the assemblages of the streams remained essentially unchanged over a 30-year period (Chester et al. 2014) suggesting drought did not have an adverse effect on populations. In another study, species distribution models were developed by Bond et al. (2011) that linked the probability of occurrence of īnanga with physiographic, climatic, and hydrologic characteristics of rivers across Victoria, in south-eastern Australia. The distribution models were then used to derive probabilities of occurrence throughout the stream network based on baseline (historic) and future climate scenarios (Bond et al. 2011). Climate scenarios corresponded to the A1T (low), A2 (median) and A1FI (high) scenarios from the Intergovernmental Panel on Climate Change (IPCC)'s Special Report on Emission Scenarios (SRES marker scenarios IPCC, 2000; see Bond et al. 2011), with associated temperature increases (by 2030) of +0.54 °C, +0.85 °C and +1.24 °C, respectively. Results indicated no changes to īnanga distributions at the lower, median or upper climate change scenario.

Īnanga can maintain basal metabolism, locomotion and feeding activity during both winter and summer which are both highly energy demanding seasons especially in the more southern parts of their distribution (Boy et al. 2017). Īnanga can be subjected to extreme cold temperatures during winter (close to their tolerance limit) and can expend energy higher than theoretically expected so that homeostasis can be maintained (Boy et al. 2017).

The distribution of īnanga spawning habitat is a function of estuary geomorphology (Hicks et al. 2010). Low salinity surface waters are extensive in stratified estuaries with concomitant extensive spawning

habitat distribution. In comparison, well-mixed estuaries have a more restricted saline wedge and spawning habitat distribution (Hicks et al. 2010). Sea-level rise and the concomitant change in the distribution of the saltwater wedge may affect the location and integrity of īnanga spawning habitats. Changes in salinity and tidal currents associated with climate change may cause significant changes in the zooplankton community structure of coastal lakes in Aotearoa-New Zealand (Hall and Burns 2003). Severe perturbations of zooplankton community structure and abundance are caused by even minor saline intrusions into Lake Waihola that raise the salinity to >1.2 ps. This indicates that relatively small increases in salinity levels can drive coastal lakes to a state of depleted biodiversity and abundance, altering ecosystem functioning (Schallenberg et al. 2003) and may alter the diet of post-larval and juvenile īnanga.

Īnanga spawning requirements are specific in terms of timing and salinity characteristics, with spawning occurring on spring tides, amongst littoral vegetation and in fresh water but just upstream of estuarine saline wedges⁷. Given spawning and recruitment are salinity-dependent, salinity changes associated with climate change may contract or fragment īnanga distributions (Booth et al. 2011).

In coastal lakes such as Lake Ellesmere, Glova and Sagar (2000) found that īnanga were distributed around the lake margin, particularly along the western side of the lake. No īnanga were caught at offshore sites, suggesting that wind would displace these fish to the lee shore by wind-driven currents (Taylor 1996). It was hypothesised that such pelagic species could become displaced by winds, but experience with frequent winds may lead fish to occupy the specific sheltered embayments along the lake margins (Crow and Bonnet 2012). Changes in the magnitude and direction of winds, especially southerlies, can have a significant effect on dispersal in other pelagic species along the west coast of the North Island (Salinas-de-Leon et al. 2012). Additionally, variation in tidal current flows can affect dispersal of species on the north-eastern coast of Aotearoa-New Zealand (Stephens et al. 2006).

3.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between īnanga and the wider ecosystem include:

- Īnanga are an important food resources for native species such as longfin and shortfin eels. They are known to constitute an important resource for kahawai (*Arripis trutta*);
- Post-spawned brown trout often migrate downstream, especially females, and estuaries afford great areas to replenish lost condition – in Canterbury, Stokells smelt is probably the main species eaten but whitebait will be also consumed;
- Kākahi (freshwater mussel; *Echyridella menziesii*) larvae are external parasites on fish during the first three weeks of life. If they do not find a fish host, they die. Kōaro are the main species cited as a host, but it is likely that all fish species, including īnanga, can be hosts;
- As a mobile consumer with distinct ontogenetic shifts in diet and habitat use, īnanga provide crucial links among production pathways of littoral, benthic and pelagic zones (Cussac et al. 1992; Barriga et al. 2002; Milano et al. 2013);
- Larval *G. maculatus* are the main planktivorous fish in Patagonian lakes (Cervellini et al. 1993). The presence of these larvae as a predator in a lake influences not only the zooplankton size structure and species composition (Modenutti et al. 1993), but also enhances the phytoplankton biomass because of excretion (Reissig et al. 2003).

3.9 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). In addition to the stressors derived from the IUCN threat assessment (n=5), which included ‘dams and water management/use [abstraction of surface and groundwater; large dams]’ and ‘invasive non-native/alien species [rainbow and brown trout]’ (see Supplementary Appendix 2) other pressures include:

- The loss of rearing and spawning habitat due to land clearance, loss of river/habitat connectivity through tide and flood gates and stop banking, channelisation, weirs, dams and land and wetland drainage has significantly reduced the distribution of this species;
- Loss of riparian vegetation (through stock trampling, grazing and mowing, for example) is a threat to spawning sites and population productivity (Hickford and Schiel 2011);
- Prior to fishery regulations, the whitebait fishery contributed to a major decline of this species;
- Predation by introduced salmonids is a potential threat to this species;
- Predation by non-native species (i.e., mice) may not be a major cause of mortality for īnanga eggs, but it might be an additional stressor on an already heavily-impacted life-history;
- The cumulative effects of spawning habitat degradation, through intensified farming, urban development and channelization, and egg predation, together with increasing fishing pressure may disrupt the balance between productive sources and population sinks (Hickford and Schiel 2010).

3.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- Degraded habitats and environmental conditions essential for spawning, survival and growth, are significant risks to this species;
- Īnanga stock structure is poorly resolved. Greater spatial coverage is needed to ascertain if more stocks exist along with a better integration of multiple techniques like genetic and otolith analyses.
- More research is required to understand recruitment trends and the impacts of whitebait harvest on īnanga populations (Egan et al. 2017);

3.11 References

Allibone, R., Boubée, J.A.T., West, D. (1999) The ones that got away: determining whitebait movements and rates of escape. *Water & Atmosphere*, 7: 11–13.

Allibone, R. (2003) Egg cannibalism by inanga (*Galaxias maculatus*). *New Zealand Journal of Marine and Freshwater Research*, 37(4): 763–765.

- Baker, C.F., Boubée, J.A.T. (2003) How many whitebait end up in our fritters? *Fishing: coast to coast*, 16: 26–27.
- Baker, C.F., Hicks, B.J. (2003) Attraction of migratory inanga (*Galaxias maculatus*) and koaro (*Galaxias brevipinnis*) juveniles to adult galaxiid odours. *New Zealand Journal of Marine and Freshwater Research*, 37: 291–299.
- Bannon H.J., Ling, N. (2004) Running the unseen, lowland gauntlet: compounding effects of temperature, hypoxia and exercise for diadromous fishes. In: Rupp GL, White MD (eds) Proceedings of the 7th international symposium on fish physiology, toxicology, and water quality, Tallinn, Estonia, 2004. U.S. Environmental Protection Agency, Ecosystems Research Division, Athens, Georgia, USA, EPA 600/R-04/0 49: 207–218.
- Bannon, H.J. (2006) Effects of Water Quality Parameters on Prolonged Swimming Ability of Freshwater Fishes. PhD thesis, University of Waikato, Hamilton, New Zealand. Retrieved from <https://hdl.handle.net/10289/2618>
- Barbee, N.C., Hale, R., Morrongiello, J., Hicks, A., Semmens, D., Downes, B.J., Swearer, S.E. (2011) Large-scale variation in life history traits of the widespread diadromous fish, *Galaxias maculatus*, reflects geographic differences in local environmental conditions. *Marine and Freshwater Research*, 62: 790–800.
- Barriga, J.P., Battini, M.A., Macchi, P.J., Milano D., Cussac, V.E. (2002) Spatial and temporal distribution of landlocked *Galaxias maculatus* and *Galaxias platei* (Pisces: Galaxiidae) in a lake in the South American Andes. *New Zealand Journal of Marine and Freshwater Research*, 36(2): 345–359.
- Barriga, J.P., Battini, M. Á., García-Asorey, M., Carrea, C., Macchi, P.J., Cussac, V.E. (2012) Intraspecific variation in diet, growth, and morphology of landlocked *Galaxias maculatus* during its larval period: the role of food availability and predation risk. *Hydrobiologia*, 679: 27–41.
- Battini, M., Rocco, V., Lozada, M., Tartarotti, B. and Zagarese, H.E. (2000) Effects of ultraviolet radiation on the eggs of landlocked *Galaxias maculatus* (Galaxiidae, Pisces) in northwestern Patagonia. *Freshwater Biology*, 44(3): 547–552.
- Becker, A., Laurenson, L.J. (2008) Seasonal and diel comparisons of the diets of four dominant fish species within the main channel and flood-zone of a small intermittently open estuary in south-eastern Australia. *Marine and Freshwater Research*, 58: 1086–1095.
- Benzie, V. (1968) Stages in the normal development of *Galaxias maculatus attenuatus* (Jenyns). *New Zealand Journal of Marine and Freshwater Research*, 2: 606–627.
- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions*, 20: 931–942.
- Booth, D.J, Bond, N., Macreadie, P. (2011) Detecting range shifts among Australian fishes in response to climate change. *Marine and Freshwater Research*, 62: 1027–1042.
- Boy, C.G., Pérez, A.F., Tagliaferro, M.B., Lattuca, M.E., Gutierrez, M.E. (2017) Exploring bioenergetics of diadromous *Galaxias maculatus* in the southernmost extreme of its

- distribution: Summer is not always the better season. *Journal of Experimental Marine Biology and Ecology*, 488: 3.
- Catlin, A.K., Collier, K.J., Duggan, I.C. (2019) Diet of juvenile *Galaxias maculatus* (Galaxiidae) during the upstream migration period in the lower Waikato River, New Zealand. *Marine and Freshwater Research*, 70: 816–823.
- Cervellini, P.M., Battini, M.A., Cussac, V.E. (1993) Ontogenetic shifts in the diet of *Galaxias maculatus* (Galaxiidae) and *Odontesthes microlepidotus* (Atherinidae). *Environmental Biology of Fishes*, 36: 283–290.
- Chapman, A. (2003) Biology of the spotted minnow *Galaxias maculatus* (Jenyns 1842) (Pisces: Galaxiidae) on the south coast of Western Australia. Masters by Research thesis, Murdoch University.
- Chapman, A., Morgan, D. L., Beatty, S. J., and Gill, H. S. (2006) Variation in Life History of Land-Locked Lacustrine and Riverine Populations of *Galaxias maculatus* (Jenyns 1842) in Western Australia. *Environmental Biology of Fishes*, 77: 21–37.
- Charteris, S.C., Allibone, R.M., Death, R.G. (2003) Spawning site selection, egg development, and larval drift of *Galaxias postvectis* and *G. fasciatus* in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, 37: 493–505.
- Chessman, B.C. (2013) Identifying species at risk from climate change: Traits predict the drought vulnerability of freshwater fishes. *Biological Conservation*, 160: 40–49.
- Chester, E. T., Matthews, T. G., Howson, T. J. , Johnston, K., Mackie, J. K., Strachan, S. R and Robson, B. J. (2014) Constraints upon the Response of Fish and Crayfish to Environmental Flow Releases in a Regulated Headwater Stream Network. *PLOS ONE*, 9: e91925.
- Crow, S., Bonnett, M. (2013) Te Waihora Mahinga Kai: a compilation of data and summary of existing research on freshwater fishes in Te Waihora. Whakaora Te Waihora Partners.
- Cussac, V.E., Cervellini, P.M., Battini, M.A. (1992) *Environmental Biology of Fish*, 35: 141. <https://doi.org/10.1007/BF00002189>
- David, B., West, D, Franklin, P., Allibone, R, Ling, N., Hitchmough, R., Crow, S. (2014) *Galaxias maculatus*. The IUCN Red List of Threatened Species 2014: e.T197279A2481214. <http://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T197279A2481214.en>
- Devictor, V., Clavel, J, Julliard, R., Lavergne, S., Mouillot, D., Thuiller, W., Venail, P., Villeger S., Mouquet, N. (2010) Defining and measuring ecological specialization. *Journal of Applied Ecology*, 47: 15–25.
- Egan, E.M. (2017) Early life history of the amphidromous galaxiid īnanga; disentangling the consequence for their migratory dynamics, population structure and adult growth. PhD. University of Canterbury, Christchurch, New Zealand.
- Egan, E.M.C., Hickford, M.J.H., Schiel, D.R. (2019) Understanding the life histories of amphidromous fish by integrating otolith-derived growth reconstructions, post-larval migrations and reproductive traits. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29: 1391–1402.

- Fraser, M.M., Lallas, C. (2004) Seasonal variation in the diet of blue penguins (*Eudyptula minor*) at Oamaru, New Zealand. *Notornis* 51: 7–15.
- Glover, C.N, Donovan, K.A., Hill, J.V. (2012) “Is the Habitation of Acidic-Water Sanctuaries by Galaxiid Fish Facilitated by Natural Organic Matter Modification of Sodium Metabolism?” *Physiological and Biochemical Zoology*, 85(5): 460–469.
- Gomon, M., Bray, D.J. (2011) Common Galaxias *Galaxias maculatus*, in Fishes of Australia. Victoria Available at: <http://fishesofaustralia.net.au/home/species/2129>. (Accessed: 22 July).
- Goodman, J.M. (2018) Conservation, ecology and management of migratory galaxiids and the whitebait fishery: a summary of current knowledge and information gaps Department of Conservation Nelson.
- Hall, C.J., Burns, C.W. (2003) Responses of crustacean zooplankton to seasonal and tidal salinity changes in the coastal Lake Waihola, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 37: 31–43.
- Harzmeyer, J.R. (2006) Effects of salinity and temperature on hatchability, and early development of *Galaxias maculatus*. Honours thesis, Deakin University, Australia.
- Hickford, M.J., Schiel, D.R. (2003) Comparative dispersal of larvae from demersal versus pelagic spawning fishes. *Marine Ecology Progress Series*, 252: 255–7271.
- Hickford, M.J.H., Cagnon, M., Schiel, D.R. (2010) Predation, vegetation and habitat-specific survival of terrestrial eggs of a diadromous fish, *Galaxias maculatus* (Jenyns, 1842). *Journal of Experimental Marine Biology and Ecology*, 385: 66–72. DOI: <https://doi.org/10.1016/j.jembe.2010.01.010>.
- Hickford, M.J.H., Schiel, D.R. (2011) Synergistic interactions within disturbed habitats between temperature, relative humidity and UVB radiation on egg survival in a diadromous fish. *PLOS ONE*, 6: e24318-e24318. DOI: 10.1371/journal.pone.0024318.
- Hickford, M.J., Schiel, D.R. (2016) Otolith microchemistry of the amphidromous *Galaxias maculatus* shows recruitment to coastal rivers from unstructured larval pools. *Marine Ecology Progress Series*, 548: 197–207.
- Hicks, A., Barbee, N. C., Swearer, S.E., Downes, B.J. (2010) Estuarine geomorphology and low salinity requirement for fertilisation influence spawning site location in the diadromous fish, *Galaxias maculatus*. *Marine and Freshwater Research*, 61: 1252–1258.
- Jellyman, P.G., Harding, J.S. (2014) Variable survival across low pH gradients in freshwater fish species. *Journal of Fish Biology*, 85: 1746–1752.
- Jowett, I.G. (2002) In-stream habitat suitability criteria for feeding inanga (*Galaxias maculatus*). *New Zealand Journal of Marine and Freshwater Research*, 36: 399–407.
- Jowett, I.G., Richardson, J. (2008) Habitat use by New Zealand fish and habitat suitability NIWA *Science and Technology Series No.55*. 148 p.

- Kaemingk, M., Swearer, S., Bury, S., Shima, J. (2019) Landscape edges shape dispersal and population structure of a migratory fish. *Oecologia*, 1–10.
- Landman, M.J, Van Den Heuvel, M.R., Ling, N. (2005) Relative sensitivities of common freshwater fish and invertebrates to acute hypoxia. *New Zealand Journal of Marine and Freshwater Research*, 39(5): 1061–1067. DOI: 10.1080/00288330.2005.9517375
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand’s diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- Marcogliese, D.J. (2008) The impact of climate change on the parasites and infectious diseases of aquatic animals. *Revue Scientifique Et Technique-Office International Des Epizooties*, 27: 467–484.
- McClintock, G.J. (2018) Early life history dynamics of the New Zealand whitebait species MSc thesis, University of Canterbury, Christchurch, New Zealand.
- McDowall, R.M. (1968) *Galaxias maculatus* (Jenyns), the New Zealand Whitebait. *New Zealand Marine Department, Fisheries Research Bulletin*, 2: 1–83.
- McDowall, R.M., Robertson, D.A., Saito, R. (1975) Occurrence of Galaxiid larvae and juveniles in the sea. *New Zealand Journal of Marine and Freshwater Research*, 9: 1–9.
- McDowall, R.M. (1990) New Zealand freshwater fishes: a natural history and guide. 2nd edition. Heinemann Reed, Auckland, New Zealand.
- McDowall, R.M. (1991) Conservation and management of the whitebait fishery. *Department of Conservation Science and Research Series*, 38: 1–18.
- McDowall, R.M., Mitchell, C.P., Brothers, E.B. (1994) Age at migration from the sea of juvenile *Galaxias* in New Zealand (Pisces, galaxiidae). *Bulletin of Marine Science*, 54: 385–402.
- McDowall, R.M. (2003) Variation in Vertebral Number in Galaxiid Fishes, How Fishes Swim and a Possible Reason for Pleomerism. *Review of Fish Biology and Fisheries*, 13: 247–263.
- McDowall, R.M. (2010) Why be amphidromous: expatrial dispersal and the place of source and sink population dynamics? *Reviews in Fish Biology and Fisheries*, 20: 87–100.
- Ministry for the Environment (2018) Climate Change Projections for New Zealand: Atmosphere Projections Based on Simulations from the IPCC Fifth Assessment, 2nd Edition. Wellington: Ministry for the Environment. <https://www.mfe.govt.nz/sites/default/files/media/Climate%20Change/Climate-change-projections-2nd-edition-final.pdf>
- Milano, D., Aigo, J.C., Macchi, P.J. (2013) Diel patterns in space use, food and metabolic activity of *Galaxias maculatus* (Pisces: Galaxiidae) in the littoral zone of a shallow Patagonian lake. *Aquatic Ecology*, 47: 277–290.
- Mitchell, C.P. (1990) Whitebait spawning grounds on the Lower Waikato River. *New Zealand Freshwater Fisheries Miscellaneous Report* 42. ? p.

- Mitchell, C., Davis-Te Maire, K. (1994) Mahinga Kai Resources of the Waiau Catchment. *Report to Electricity Corporation of New Zealand*: 130.
- Modenutti, B.E., Balseiro, E.G., Cervellini, P.M. (1993) Effect of the selective feeding of *Galaxias maculatus* (Salmoniformes, Galaxiidae) on zooplankton of a South Andes lake. *Aquatic Sciences*, 55: 65–75.
- Neilson, C.S.B. (2016) There and back again; spatial and temporal variation in the recruitment dynamics of an amphidromous fish. MSc thesis, Victoria University, Wellington, New Zealand.
- Olsen, D.A., Tremblay, L., Clapcott, J., Holmes, R. (2012) Water temperature criteria for native aquatic biota. *Auckland Council Technical Report 2012/036*.
- Pollard, D. (1972) The biology of a landlocked form of the normally catadromous salmoniform fish *Galaxias maculatus* (Jenyns). III. Structure of the gonads. *Marine and Freshwater Research*, 23: 17–38.
- Reissig, M., Queimalinos, C.P., Balseiro, E.G. (2003) Effects of *Galaxias maculatus* on nutrient dynamics and phytoplankton biomass in a North Patagonian oligotrophic lake. *Environmental biology of fishes*, 68: 15–24.
- Richardson, J., Taylor, M.J. (2002) Reprinted with minor revisions 2004 A guide to restoring īnanga habitat. *NIWA Science and Technology Series No. 50*: 31.
- Rowe, D.K., Kelly, G. (2009) Duration of the oceanic phase for īnanga whitebait (Galaxiidae) is inversely related to growth rate at sea. Pages 343-354. In A. Haro, K.L. Smith, R.A. Rulifson, C.M. Moffitt, R.J. Klauda, M.J. Dadswell, R.A. Cunjak, J.E. Cooper, K.L. Beal, T.S. Avery, editors. Challenges for diadromous fishes in a dynamic global environment. American Fisheries Society, Halifax, Canada.
- Ruiz-Jarabo, I., González-Wevar, C. A., Oyarzún, R., Fuentes, J., Poulin, E., Bertrán, C. and Vargas-Chacoff, L. (2016) Isolation Driven Divergence in Osmoregulation in *Galaxias maculatus* (Jenyns, 1848) (Actinopterygii: Osmeriformes). *PLOS ONE*, 11: e0154766-e0154766.
- Salinas-de-León, P., Jones, T., Bell, J.J. (2012) Successful determination of larval dispersal distances and subsequent settlement for long-lived pelagic larvae. *PLOS ONE*, 7: e32788.
- Schallenberg, M., Hall, C.J., Burns, C.W. (2003) Consequences of climate-induced salinity increases on zooplankton abundance and diversity in coastal lakes. *Marine Ecology Progress Series*, 251: 181–189.
- Simons, M. (1986) Effects of elevated temperatures on two whitebait species, *Galaxias maculatus* and *G. fasciatus*, and the shortfinned eels, *Anguilla australis*. *Technical Publication, Waikato Valley Authority*: 62.
- Stancliff, A.G., Boubée, J.A.T., Mitchell, C.P. (1988a) The whitebait fishery of the Waikato River. *New Zealand Freshwater Fisheries Report*, 95: 68.

- Stancliff, A.G., Boubée, J.A.T., Palmer, D., Mitchell C.P. (1988b) The upstream migration of whitebait species in the lower Waikato River. *New Zealand Freshwater Fisheries Report* 96: 44. NIWA, Rotorua.
- Stephens, S.A., Broekhuizen, N., Mac diarmid, A.B., Lundquist, C.J., McLeod, L., Haskew, R. (2006) Modelling transport of larval New Zealand abalone (*Haliotis iris*) along an open coast. *Marine and Freshwater Research*, 57: 519–532.
- Stevens, J.C.B., Hickford, M.J.H., Schiel, D.R. (2016) Evidence of iteroparity in the widely distributed diadromous fish inanga *Galaxias maculatus* and potential implications for reproductive output. *Journal of fish biology*, 89: 1931–1946.
- Taylor, M. (1996) The Natural Resources of Lake Ellesmere (Te Waihora) and its Catchment, Environment Canterbury, Christchurch, New Zealand. Report 96: 322.
- Urbina, M.F. (2013) Strategies of inanga (*Galaxias maculatus*) for surviving the environmental stressors of hypoxia and salinity change. PhD thesis, University of Canterbury.
- Waters, J.M., Dijkstra, L.H., Wallis, G.P. (2000) Biogeography of a southern hemisphere freshwater fish: how important is marine dispersal? *Molecular Ecology*, 9: 1815–1821.
- Wedderburn, S.D., Barnes, T.C, Hillyard, K.A. (2014) Shifts in fish assemblages indicate failed recovery of threatened species following prolonged drought in terminating lakes of the Murray–Darling Basin, Australia. *Hydrobiologia*, 730: 179–190.
- Wood, V.B.T. (2016) Three rivers and a fish: body shape variation in adult inanga (*Galaxias maculatus*) and implications for reproductive output. MSc. Victoria University Wellington, New Zealand.
- Yungnickel, M.R. (2017) New Zealand's whitebait fishery: spatial and temporal variation in species composition and morphology. MSc thesis, University of Canterbury, Christchurch, New Zealand.

4 Kōaro (*Galaxias brevipinnis*)



Figure 4-1: Adult kōaro (*Galaxias brevipinnis*). Source: NIWA.

4.1 Habitats

Kōaro are found in six different habitat types over their life cycle (see Supplementary Appendix 2). Although widespread around the country, adult kōaro (Figure 4-1) typically prefer clear, swiftly flowing streams of small to moderate size that flow through native forest. Kōaro are noted climbers and are found in the headwaters of many catchments around Aotearoa-New Zealand. Kōaro can occupy tussock streams, particularly those flowing into high-altitude lakes.

This species commonly forms extensive lacustrine populations in Aotearoa-New Zealand. Lacustrine (lake) populations can reach very high densities of individuals due to high recruitment success (David et al. 2014). During the summer and autumn, landlocked adult kōaro emigrate into inlet springs and streams to spawn in lake tributaries (Rowe et al. 2002). The larvae are pelagic, can be found at variable depths and likely move to littoral (surface) zone when 30-35 mm in length (Goodman 2018). Juvenile lacustrine kōaro likely occupy open surface waters but can also be found in lake margins. They can either migrate into lake tributaries or remain in lakes (Goodman 2018). Kōaro were sampled from four areas above lakes in the eastern South Island biogeographical region, and their reproductive traits measured (Jones et al. 2016). Kōaro showed relatively limited intraspecific trait variation (i.e., differences in their traits among populations), which indicates that habitat specialisation occurs and that their life-history constraints may prevent kōaro from colonising a wider variety of habitats (Jones et al. 2016).

Aotearoa-New Zealand and Australian populations of this species are likely to be split into separate species in the future, as genetic data indicate significant diversity between them (Waters and Wallis 2001) and four species likely exist in Australia (Raadik et al. 2019). The Aotearoa-New Zealand species will remain as *G. brevipinnis*, as it was originally described (David et al. 2014), but a revision of its status will be needed once this species complex is resolved and taxa described (Raadik et al. 2019). In Aotearoa-New Zealand, genetic analysis indicates that there is enough mixing of individuals between rivers and 'stocks' to ensure that speciation is not occurring (J. Goodman, DOC, unpubl. data). However, continual loss of habitat creating greater fragmentation of 'stocks' may reduce mixing and therefore genetic exchange in the future (Goodman et al. 2018).

4.2 The fishery

The juvenile life-stage is a component of the Aotearoa-New Zealand whitebait fishery. Peak migration for kōaro is considered to be September (Goodman 2018). At specific times of the year, and in certain rivers/regions, kōaro can comprise a significant proportion of the whitebait catch (Figure 4-2). The causes of kōaro decline includes general range (distribution) contraction and decrease in the abundance of large galaxiids, in part due to the over-harvesting of the juvenile whitebait stage (David et al. 2014).

Research to better understand kōaro stock structure(s) with respect to whitebait fisheries management have not been completed. However, there is preliminary evidence for spatial differences in size and age at migration that show similar patterns to īnanga (McDowall et al. 1994; Yungnickel 2017). Resolving the stock structure of kōaro is important to ensure the sustainability of the whitebait fishery, especially considering kōaro can comprise up to 25% of the whitebait catch in some parts of Aotearoa-New Zealand (Figure 4-2, Yungnickel 2017).

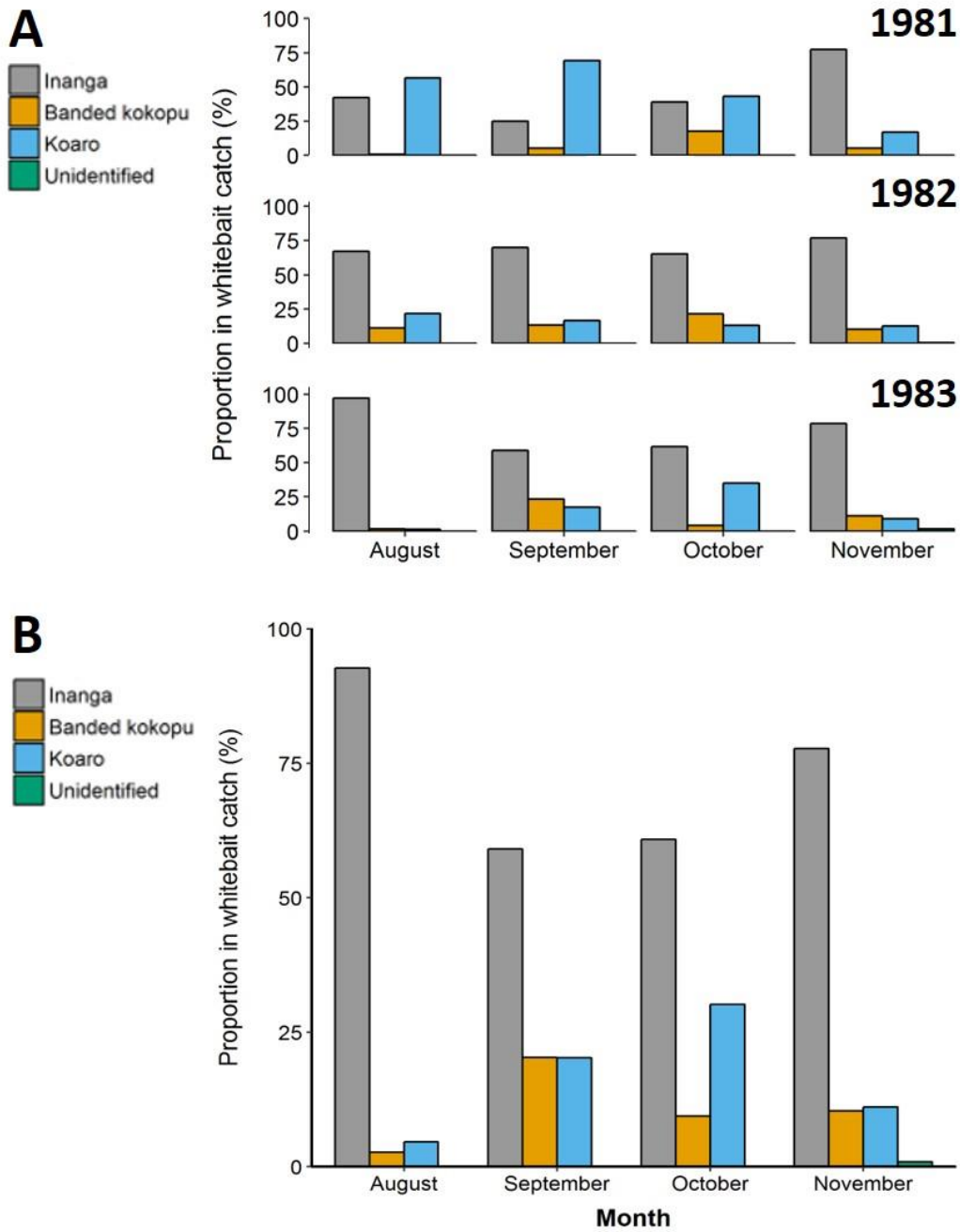


Figure 4-2: (A) Monthly variation in whitebait species composition across 12 Bay of Plenty rivers between 1981-1983; and (B) Average monthly variation in species composition between 1981-1983 for 12 Bay of Plenty rivers. Source: Redrawn by Egan from data in Rowe et al. (1992).

In New Zealand, trends in the relative abundance of whitebait are limited because of the lack of consistently collected data. Crow et al. (2016) completed an analysis of temporal trends in freshwater fish abundance using the New Zealand Freshwater Fish Database (NZFFD) from data collected between 1977–2015. Kōaro was the only galaxiid species with adequate data to enable an assessment to be carried out. Crow et al. (2016) calculated two trend metrics over the 1977–2015 period for kōaro, one of these metrics suggested that the relative abundance has stayed the same over this time period (Figure 4-3) while the other trend metric suggested that the relative abundance of kōaro was decreasing at a median ($\pm 95\%$ CI) rate of 0.05 (± 0.02) %/year (Figure 4-2). Globally, kōaro are thought to be experiencing localised population declines in some parts of its range (Raadik et al. 2019). However, the rate of decline has slowed significantly, and the species is considered widely distributed throughout its current range (Raadik et al. 2019).

4.3 Life cycle, age and growth

Kōaro is considered a long-lived species (more than 15 years age) that breed every year from about three years of age, with a generation length of eight years (McDowall 2000; David et al. 2014). Kōaro deposit their eggs amongst marginal gravels and litter during periods of elevated stream flow (Allibone and Caskey 2000) but spawning has also been found on macrophytes. Kōaro larvae hatch typically 3–4 weeks later if the eggs are re-inundated during high flow events. The hatched larvae (about 7–8 mm long) go to sea to feed and grow for about 17–20 weeks, then as whitebait (c. 45–50 mm long) migrate upstream in early spring (McDowall 1990). Spawning sites have been recently located in riffle habitat on the underside of boulders (see references in Goodman 2018).

In riverine populations, kōaro spawning occurs during autumn/winter. Downstream drift of kōaro larvae was observed in May by Charteris et al. (2003) for Taranaki streams, but kōaro larvae have also been observed in March in South Island streams (McDowall and Suren 1995). McDowall and Suren (1995) reported downstream migrating larval kōaro in February in the Otira River and concluded that the spawning habitat occurred instream rather than on riparian vegetation, with spawning and hatching occurring independent of floods. Examination of hatch date distributions of returning kōaro whitebait to South Island rivers observed a spawning season of May through to July (McDowall et al. 1994) while a more recent study showed that the larvae were hatched from April through to July (McClintock 2018). Landlocked populations reproduce mostly in winter (Jones et al. 2016). There is no relationship between egg size and female body size, but larger females produce more eggs (Jones et al. 2016).

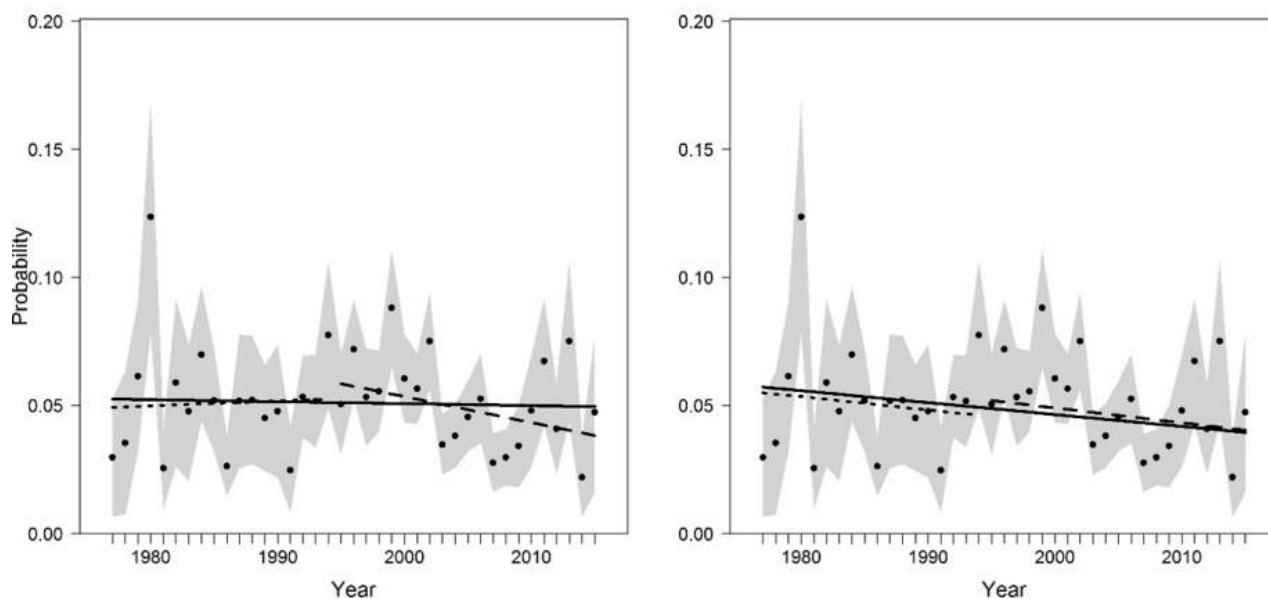


Figure 4-3: Changes in the probability of capture of kōaro through time (1970–2015). Crow et al. (2016) used data from the NZFFD to generate trends in the relative abundance of kōaro. Plots show the characteristic probability of capture for each year (black circles) and 95% CI (grey shaded area). To identify if ‘probability of capture’ through time is increasing (getting better) or decreasing (getting worse) Crow et al. (2016) used two different techniques, Sen Slope Estimator (left) and Weighted Sen Slope Estimator (right). The results of these analyses are shown for 1977–2015 (solid black line), 1977–1994 (dotted black line) and 1995–2015 (dashed black line). Source: Crow et al. (2016).

For landlocked populations, kōaro have a protracted ripening and spawning time compared with diadromous populations (Augspurger and Closs 2019). A recent study found that diadromous populations of kōaro simultaneously spawned during a March flood, whereas ovulating landlocked individuals were found from August to December, with a peak in October (Augspurger and Closs 2019). Oocyte size in landlocked individuals was approximately 25% smaller than that in diadromous populations, and the fecundity of landlocked individuals was higher than that for diadromous individuals (Augspurger and Closs 2019). Recent studies show that kōaro larvae in lakes display strong signals to flows (Dr Jason Augspurger, pers. comm.), meaning that the dispersal of larvae is likely influenced by flows, but at present this is not well understood. Key demographic parameters are shown in Table 4-1.

Table 4-1: Key demographic characteristics for kōaro. Where: *denotes estimates from North Island populations and + from South Island populations.

Key demographic parameters	Estimates	Reference
Maximum size (mm)	300	McDowall (1990)
Minimum size sexual maturity (mm)	–	–
Age at sexual maturity (yrs)	–	–
Longevity (yrs)	208.0 mm (Female) = 8 yrs* 207.0 mm (Male) = 6+ yrs*	David (1989)

Key demographic parameters	Estimates	Reference
Growth parameter (K)	–	–
Fecundity	6896 (210 mm Female) 2400 (± 414)	NIWA unpubl. Data Jones et al. (2017)
Larval size _{hatch} (mm)	8.46 (mean; range = 7.6-9.0)	McDowall and Suren (1995)
Egg size (mean)	1.54	NIWA unpubl. data; Allibone and Caskey 2000
Generation length	8 years	David et al. (2014)

4.4 Distribution and environmental preferences

Kōaro are found throughout Aotearoa-New Zealand (including Chatham and Stewart Islands and on the sub-Antarctic Auckland and Campbell Islands: Figure 4-4). The natural distribution of this species is fragmented throughout its range (probably due to habitat loss and degradation). However, kōaro can migrate to more remote places (usually less impacted by anthropogenic pressures including introduced predatory fish) (McDowall 1990) because of their skilled climbing abilities. Kōaro are less common on the east coast of the South Island but can easily be found in areas of suitable habitats and are usually at higher altitudes.

In the upper North Island, kōaro can be found close to the coast and at much lower altitudes compared to South Island populations that tend to be found in higher altitude environments much further inland (David et al. 2014). Kōaro can be abundant in the tributary streams of many deep South Island lakes at varying altitudes from sea level on the West Coast to higher altitudes at multiple locations (David et al. 2014). Kōaro have commonly formed extensive lacustrine populations in many of Aotearoa-New Zealand's inland lakes and alpine tarns including artificially enclosed water bodies (i.e., reservoirs). Kōaro are thought to form landlocked populations in colder, less productive lakes, indicating that kōaro larvae could be limited by temperature, but this could also be associated with productivity and food abundance (also linked to temperature; see reference in David et al. 2014). The environmental variables associated with kōaro distribution include rain days (Table A-1; Leathwick et al. 2008).

Localised extinctions of kōaro have occurred in Lake Rotopounamu and are attributed to the introduction of smelt (*Retropinna retropinna*; Rowe 1993). Interspecific competition for food between 0+ year old kōaro and smelt, combined with predation by 2+ year old smelt on kōaro larvae, are thought to be responsible. Rainbow trout (*Oncorhynchus mykiss*) also predate on kōaro but this predation pressure has not resulted in extinctions although trout almost decimated kōaro populations in Lake Taupō.

No specific information on the distribution of kōaro larvae during marine development exists. However, galaxiid larvae more generally are found in the marine plankton year-round, up to 250 km offshore (McDowall et al. 1975) but are most abundant up to 6 km offshore (Hickford and Schiel 2003). It is suspected that larval biology / ecology plays a major role in determining kōaro distribution, but as we know relatively little about larval biology and ecology it is not possible to be conclusive (Gerry Closs, pers. comm. 2014).



Figure 4-4: Kōaro distribution in Aotearoa-New Zealand. Locations of NZFFD records where kōaro are present (black circles) and absent (grey circles). Source: NIWA.

4.5 Diet and predation

Only a handful of studies on kōaro diet exist. Kōaro are generalists and eat both aquatic and terrestrial organisms. In lakes, kōaro diet was dominated, both numerically and by weight, by aquatic prey: Ephemeroptera, Trichoptera, and Diptera larvae (Kusabs and Swales 1991). Terrestrial prey items in their diet were present in low numbers but were more important in terms of weight. Compared to the diet of rainbow trout, resource partitioning was weak although kōaro consumed more small benthic invertebrates such as chironomid larvae, whereas the diet of rainbow trout contained more Ephemeroptera larvae and terrestrial insects. Adult kōaro also consume smaller kōaro in lakes, while in streams, kōaro are known to consume both rainbow trout ova and kōaro ova (Kusabs and Swales 1991). In a large braided river system, *Deleatidium* larvae, and *Hydrobiosis* larvae are an important component of the kōaro diet (Sagar and Eldon 1983).

The diets of kōaro and juvenile rainbow trout in some Taupō tributaries are similar, and it is thought that populations may co-exist by temporal and/or spatial partitioning of food resources, whereas trout predation on small kōaro may be a limiting factor for kōaro populations (Kusabs and Swales 1991). John Hayes wrote a report for Doc about drift feeding by koarao : see <https://www.doc.govt.nz/documents/science-and-technical/casn127.pdf>

4.6 Recruitment

Little is known about the cues kōaro use for inward migration to fresh waters, but migration is likely triggered by seasonal changes in water temperature and day length (Barbee et al. 2011) along with flood flows (McDowall 1995). Olfactory cues from other migratory galaxiids help post-larvae to select a river to return to (Baker and Hicks 2003). Rowe et al. (1992) also found evidence for stream selection in kōaro whitebait within Bay of Plenty rivers, suggesting that adult pheromones may also be used as a migration cue by this species.

Baker and Smith (2015) examined the influence of river flow on recruitment and capture rates of whitebait but species-specific relationships were not derived. It was found that increasing river flow was positively correlated with the recapture rates of whitebait. This supports the earlier study of McDowall and Eldon (1980) who suggested that larger whitebait catches would be expected after floods when river flow is high and the river is more turbid. In addition, Baker and Smith (2015) found that recapture rates were positively correlated with increasing water velocities around the edge of the whitebait trap.

Little is known specifically about the swimming behaviour of kōaro post-larvae and juveniles. Juvenile galaxiids prefer low velocity waters for migration (less than 0.1 m s^{-1} ; McDowall and Eldon 1980), and fish often move upstream in the low velocity surface waters ($\leq 1.0 \text{ m}$ deep) along riverbank margins (Stancliff et al. 1988). As tidal waters recede, river flows produce stronger water velocities around whitebait traps for fish to negotiate. Baker and Smith (2015) found that once water velocities around fishermen's traps exceeded 0.1 m s^{-1} , marked increases in whitebait recapture rates were seen. Higher flows are generally associated with higher turbidity (Hicks et al. 2004) and this could also reduce the ability of fish to visually detect the trap, leading to increased catches.

4.7 Current impacts of climate change

Changes in flow and water temperatures are considered significant threats to kōaro (David et al. 2014) as well as severe weather patterns (Raadik et al. 2019) but overall little is known currently about

climate change impacts on this species. The probability of kōaro occurrence was linked with physiographic, climatic, and hydrologic characteristics of rivers across Victoria, in south-eastern Australia, using species distribution models (Bond et al. 2011). The distribution models were then used to derive probabilities of occurrence throughout the stream network based on baseline (historic) and future climate scenarios (Bond et al. 2011). Climate scenarios corresponded to the A1T (low), A2 (median) and A1FI (high) scenarios from the Intergovernmental Panel on Climate Change (IPCC)'s Special Report on Emission Scenarios (SRES marker scenarios IPCC, 2000; see Bond et al. 2011), with associated temperature increases (by 2030) of +0.54°C, +0.85 °C and +1.24 °C, respectively. Results indicated no change in kōaro distributions at the lower (+0.54°C), median (+0.85 °C) and upper (+1.24°C) climate change scenarios.

In another study, drought vulnerability of 43 freshwater fish species was done using 14 traits for each species (e.g., fecundity, adult size, spawning temperature; Chessman 2013). In the analysis, kōaro were ranked as the 11th most drought vulnerable freshwater fish species in Australia (Chessman 2013).

4.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between kōaro and the wider ecosystem include:

- Interspecific competition for food between 0+ year old kōaro and smelt has been observed as has predation by 2+ year old smelt and rainbow trout on kōaro larvae;
- Kōaro are generalists and eat both aquatic and terrestrial organisms. Adult kōaro also consume smaller kōaro in lakes, while in streams, kōaro are known to consume both rainbow trout ova and kōaro ova. The diets of kōaro and juvenile rainbow trout in some Taupō tributaries are similar. Trout predation on small kōaro may be a limiting factor for some kōaro populations (Kusabs and Swales 1991);
- Kākahi (freshwater mussel; *Echyridella menziesii*) larvae are external parasites on fish during the first three weeks of life. If they do not find a fish host, they die. Kōaro are one of the main species cited as a host;
- Kōaro are noted climbers and are found in the headwaters of many catchments around Aotearoa-New Zealand. Kōaro can also form landlocked populations;
- As a mobile consumer with distinct ontogenetic shifts in diet and habitat use, kōaro provide crucial links among production pathways of littoral, benthic and pelagic zones.

4.9 Additional (multiple stressors)

The stressors derived from the IUCN threat assessment (n=7), included 'fishing and harvesting of aquatic resources [harvest]'; 'dams and water management/use [abstraction of surface water and large dams]'; 'invasive non-native/alien species [rainbow and brown trout]', and 'climate change and severe weather [habitat shortfin and alteration]' (see Supplementary Appendix 2).

4.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- Kōaro stock structure is poorly resolved. Greater spatial coverage is needed to ascertain if more stocks exist along with a better integration of multiple techniques like genetic and otolith analyses;
- Significant knowledge gaps exist for key demographic parameters (Table 4-1).

4.11 References

- Allibone, R.M., Caskey, D. (2000) Timing and habitat of kōaro (*Galaxias brevipinnis*) spawning in streams draining Mt Taranaki, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 34(4): 593–595, DOI: 10.1080/00288330.2000.9516960
- Augspurger, J.M., Closs, G.P. (2019) Early life-history adaptation influences conservation approaches for facultatively amphidromous fish. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 29(9): 1403–1408.
- Baker, C.F., Hicks, B.J. (2003) Attraction of migratory inanga (*Galaxias maculatus*) and koaro (*Galaxias brevipinnis*) juveniles to adult galaxiid odours. *New Zealand Journal of Marine and Freshwater Research*, 37: 291–299.
- Baker, C.F., Smith, J.P. (2015) Influence of flow on the migration and capture of juvenile galaxiids in a large river system. *New Zealand Journal of Marine and Freshwater Research*, 49: 51–63.
- Barbee, N.C., Hale, R., Morrongiello, J., Hicks, A., Semmens, D., Downes, B.J., Swearer, S.E. (2011) Large-scale variation in life history traits of the widespread diadromous fish, *Galaxias maculatus*, reflects geographic differences in local environmental conditions. *Marine and Freshwater Research* 62: 790–800.
- Bell, C.P. (2001) The ecology of Kōaro (*Galaxias brevipinnis*) in Manson Creek, North Canterbury. MSc. University of Canterbury, Christchurch, New Zealand.
- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions* 20: 931–942.
- Booth, D.J., Bond, N., Macreadie, P. (2011) Detecting range shifts among Australian fishes in response to climate change. *Marine and Freshwater Research* 62: 1027-1042.
- Boubée, J.A.T., West, W.D., Mora, A.L. (1992) Awakino River Whitebait Fishery. Waikato Fisheries Consultants, MAF Fisheries, Hamilton, 129: 1–60.
- Charteris, S.C., Allibone, R.M., Death, R.G. (2003) Spawning site selection, egg development, and larval drift of *Galaxias postvectis* and *G. fasciatus* in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, 37: 493–505
- Chessman, B.C. (2013) Identifying species at risk from climate change: Traits predict the drought vulnerability of freshwater fishes. *Biological Conservation*, 160: 40–49.
- Crow, S., Snelder, T., Jellyman, P., Greenwood, M., Dunn, A. (2016) Temporal trends in the relative abundance of New Zealand freshwater fishes: analysis of New Zealand Freshwater Fish Database records. *NIWA Client Report: CHC2016-049*: 71

- David, D.W. (1989) The ecology of native and introduced fish in some Waikato streams. Biological Sciences. MSc. University of Waikato, Hamilton, New Zealand.
- David, B., David, D., Franklin, P., Crow, S., Ling, N., Allibone, R., Hitchmough, R. (2014b) *Galaxias brevipinnis*. *The IUCN Red List of Threatened Species 2014*: e.T197277A2481005. <http://dx.doi.org/10.2305/IUCN.UK.20143.RLTS.T197277A2481005.en>. Downloaded on 06 January 2018.
- Department of Conservation (2005) New Zealand large galaxiid recovery plan, 2003–13 Shortjaw kokopu, giant kokopu, banded kokopu, and kōaro. *Threatened Species Recovery Plan*, 55: 34.
- Egan, E., Gee, E., Baker, C. (2018) An assessment of the Waikato River whitebait fishery with potential management options. Waikato Raupatu River Trust. *NIWA Client Report 2018036HN*.
- Goodman, J.M. (2018) Conservation, ecology and management of migratory galaxiids and the whitebait fishery: a summary of current knowledge and information gaps Department of Conservation Nelson.
- Glova, G.J., Sagar, P.M. (1989) Feeding in a nocturnally active fish, *Galaxias brevipinnis*, in a New Zealand stream. *Marine and Freshwater Research*, 40: 231–240.
- Hickford, M.J., Schiel, D.R. (2003) Comparative dispersal of larvae from demersal versus pelagic spawning fishes. *Marine Ecology Progress Series*, 252: 255–271.
- Hicks, M., Quinn, J., Trustrum, N. (2004) Sediment load and organic matter. In: Hardin, J.S., Mosley, M.P., Pearson, C.P., Sorrell, B.K. eds. *Freshwaters of New Zealand*. Wellington, New Zealand. *New Zealand Hydrological Society and New Zealand Limnological Society*, 12.1–12.16.
- Hicks, A.S., Jarvis, M.G., David, B.O., Waters, J.M., Norman, M.D., Closs, G.P. (2017) Lake and species-specific patterns of non-diadromous recruitment in amphidromous fish: the importance of local recruitment and habitat requirements. *Marine and Freshwater Research*, 68: 2315–2323.
- Jones, P.E., Augspurger, J., Closs, G.P. (2017) Landscape-scale life-history gradients in New Zealand freshwater fish. *Freshw Biol*, 62: 570–581.
- Kusabs I.A., Swales, S. (1991) Diet and food resource partitioning in kōaro, *Galaxias brevipinnis* (Günther), and juvenile rainbow trout, *Oncorhynchus mykiss* (Richardson), in two Taupo streams, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 25: 317–325. DOI: [10.1080/00288330.1991.9516485](https://doi.org/10.1080/00288330.1991.9516485)
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- McDowall, R.M., Robertson, D.A., Saito, R. (1975) Occurrence of Galaxiid larvae and juveniles in the sea. *New Zealand Journal of Marine and Freshwater Research*, 9: 1–9.

- McDowall, R.M. (1990) *New Zealand freshwater fishes: a natural history and guide*. 2nd edition. Heinemann Reed, Auckland, New Zealand.
- McDowall, R.M., Mitchell, C.P., Brothers, E.B. (1994) Age at migration from the sea of juvenile *Galaxias* in New Zealand (Pisces, galaxiidae). *Bulletin of Marine Science*, 54: 385–402.
- McDowall, R.M. (1995) Seasonal pulses in migrations of New Zealand diadromous fish and the potential impacts of river mouth closure. *New Zealand Journal of Marine and Freshwater Research*, 29: 517–526.
- McDowall, R.M., Suren, A.M. (1995) Emigrating larvae of koaro, *Galaxias brevipinnis* Günther (Teleostei: Galaxiidae), from the Otira River, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 29: 271–275.
- McClintock, G.J. (2018) Early life history dynamics of the New Zealand whitebait species MSc thesis, University of Canterbury, Christchurch, New Zealand.
- Raadik, T., Bice, C., David, B., David, D, Franklin, P., Crow, S., Ling, N., Allibone, R., Hitchmough, R. (2019) *Galaxias brevipinnis*. The IUCN Red List of Threatened Species 2019: e.T197277A129040345. <http://dx.doi.org/10.2305/IUCN.UK.2019-3.RLTS.T197277A129040345.en>
- Rowe, D.K., Saxton, B.A., Stancliff, A.G. (1992) Species composition of whitebait (Galaxiidae) fisheries in 12 Bay of Plenty rivers, New Zealand: evidence for river mouth selection by juvenile *Galaxias brevipinnis* (Günther). *New Zealand Journal of Marine and Freshwater Research*, 26: 219–228.
- Rowe, D.K. (1993) Disappearance of kōaro, *Galaxias brevipinnis*, from Lake Rotopounamu, New Zealand, following the introduction of smelt, *Retropinna retropinna*. *Environmental Biology of Fishes*, 36: 329–336.
- Rowe, D., Konui, G., Christie, K. (2002) Population structure, distribution, reproduction, diet, and relative abundance of koaro (*Galaxias brevipinnis*) in a New Zealand lake. *Journal of the Royal Society of New Zealand*, 32: 275–291.
- Sagar, P.M., Eldon G.A. (1983) Food and feeding of small fish in the Rakaia River, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 17: 213–226, DOI: 10.1080/00288330.1983.9515999 To link to this article: <https://doi.org/10.1080/>
- Saxton, B.A., Rowe, D.K., Stancliff, A.G. (1987) Species composition and relative importance of whitebait fisheries in 13 Bay of Plenty rivers. *New Zealand Ministry of Agriculture and Fisheries, Fisheries Environmental Report*, 79: 1–63.
- Stancliff, A.G., Boubée, J.A.T., Palmer, D., Mitchell C.P. (1988) The upstream migration of whitebait species in the lower Waikato River. *New Zealand Freshwater Fisheries Report*, 96: 44. NIWA, Rotorua.
- Unwin, M., Davis, S.F. (1983) Recreational fisheries of the Rakaia River. *New Zealand Ministry of Agriculture and Fisheries*. Christchurch, New Zealand.
- Waters, J.M., Wallis, G.P. (2001) Cladogenesis and loss of the marine life-history phase in freshwater galaxiid fishes (Osmeriformes: Galaxiidae). *Evolution*, 55: 587–597.

Watson, J.R., Goodrich, H.R., Cramp, R.L., Gordos, M.A., Yan, Y., Ward, P.J., Franklin, C.E. (2019) Swimming performance traits of twenty-one Australian fish species: a fish passage management tool for use in modified freshwater systems. *bioRxiv*, 861898. DOI: <https://doi.org/10.1101/861898>

Yungnickel, M.R. (2017) New Zealand's whitebait fishery: spatial and temporal variation in species composition and morphology. MSc thesis, University of Canterbury, Christchurch, New Zealand.

5 Piharau / Kanakana / Pouched Lamprey (*Geotria australis*)



Figure 5-1: Pouched lamprey (*Geotria australis*). The velasia life stage (pre-reproductive adult) is pictured here (see Figure 5-2). Source: NIWA.

5.1 Habitats

The pouched lamprey (Figure 5-1) is found in four habitat types according to the IUCN threat assessment (Bice et al. 2019; see Supplementary Appendix 2). Lamprey are found in one marine neritic habitat (estuaries), two marine/oceanic habitats (Epipelagic [0-200 m] and Mesopelagic [200-1000 m]) and one wetland habitat type (Permanent Rivers/Streams/Creeks [includes waterfalls]).

In fresh water, the spawning habitats of lamprey are not well known, and observations only exist for a handful of rivers. Lamprey spawning habitat is usually found in stream sections containing large substrates with fish forming a pair and spawning beneath large boulders and bedrock (Baker et al. 2017; Table 5-1). Lamprey nesting and spawning sites have been identified in Kinloch Stream (Banks Peninsula) and the Waikawa River (Southland). These are the first spawning sites to have been confirmed in Aotearoa-New Zealand (Figure 5-2). Although sample sizes were small, the nesting habitat used by lamprey was consistent across 2 years (2013 and 2015; Baker et al. 2017), suggesting that the spawning sites are repeatedly used (i.e., spawning site fidelity).

Larval lamprey (ammocoetes) (Figure 5-3) are found burrowed in silty/sandy substrates in backwaters or along stream or river margins (McDowall 1990). The ammocoetes choose substrate on the basis of underlying, not surficial, particle size and substrate depth (Kelso 1993). Ammocoetes are associated with run (flowing water) habitat types, overhead shade, and substrate features (depth or the proportion of fine sand). The amount of particulate organic matter is negatively correlated with ammocoete abundance (Jellyman et al. 2002a). Macrophthalmia (Figure 5-3) migrate downstream into the sea, mostly during autumn and winter. During their marine life stage, they attach onto the body of other fish and to marine mammals and live as parasites.

Table 5-1: Key physical habitat features of six nesting sites for nesting and maturing *Geotria australis* (n = 18) in Okuti River and Kinloch Stream. The substrate index is based on the modified Wentworth particle size scale and ranges from vegetation (1) to bedrock (8). The nesting sites were located under significantly larger boulders than those beneath which maturing lamprey reside. Source: Baker et al. (2017).

Habitat features	Nesting fish		Maturing fish	
	Mean	Max	Mean	Max
Boulder size (length, m)	0.70	0.83	0.49	0.75
Boulder size (width, m)	0.44	0.55	0.32	0.53
Water depth (m)	0.10	0.20	0.13	0.32
Water velocity (m s ⁻¹)	0.20	0.44	0.18	0.45
Substrate index	6.50	6.7	6.17	6.95
Wetted width (mm)	2.14	2.75	2.38	4.00

Adult habitat preferences are not well described. Summer stream temperature was the only environmental variable correlated with lamprey distribution at >5% predictive power (Leathwick et al. 2008). During the day, lampreys are generally associated with bankside debris cover (Jellyman et al. 2002b). Pools are the habitat most frequently used by adult lamprey, followed by riffles and runs. Depth of water used by lampreys (n = 13 observations) ranged from 0.2 to 1.5 m (mean 0.7 m). The average velocity of locations where nine lamprey were specifically located was 0.1 m s⁻¹ (range <0.1-0.3 m s⁻¹) whereas velocities 0.1 m "outside" these sites (i.e., 0.1 m towards the stream centre) averaged 0.5 m s⁻¹ (range 0.1-0.9 m s⁻¹; Jellyman et al. 2002b).

Adult lamprey can cohabit at the same site, or the same site may be used on different occasions, and even in different years (Jellyman et al. 2002b). Once in fresh water, large flood events can delay further upstream movement and can displace some lamprey downstream. However, during periods of flood recession migration upstream can resume (Jellyman et al. 2002b).



Figure 5-2: Examples of the type of headwater stream habitat spawning lamprey seem to prefer in Kinloch Stream. The yellow arrow indicates boulders where lamprey spawning nests were located underneath. Source: NIWA.

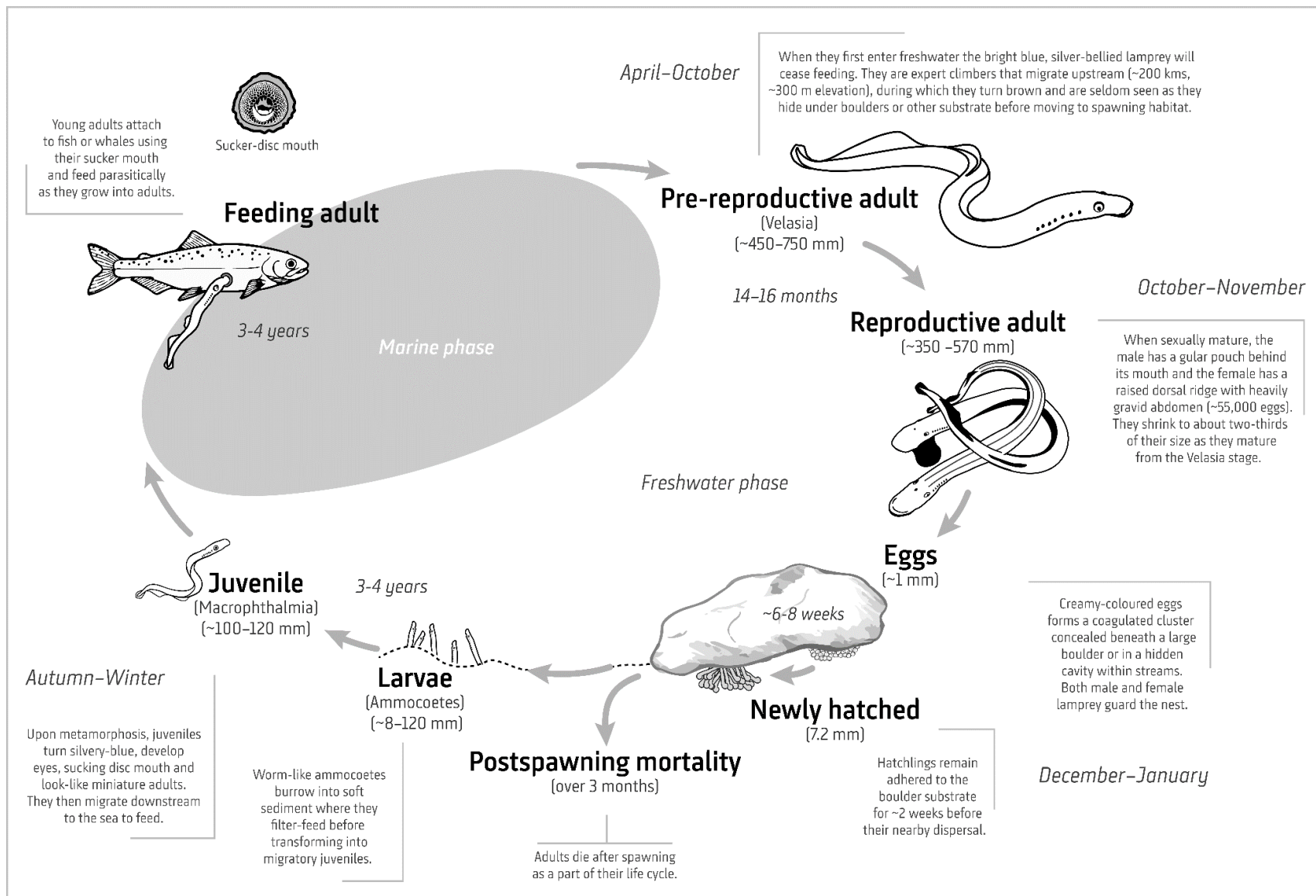


Figure 5-3: Pouched lamprey (*Geotria australis*) life cycle. Source: NIWA.

5.2 The fishery

No commercial or recreational fishery for lamprey exists in Aotearoa-New Zealand and there are no conservation actions in place for this species (Closs et al. 2014). They are customary species and still harvested in places such as Whanganui (Pipiriki) and Mataura Falls, Waikawa. There is a daily bag limit of 30 in Southern and Fiordland areas, through the Amateur Fishing Regional Fisheries (Southland and Sub-Antarctic Areas Amateur Fishing) Regulations 1991 (MPI undated). The Ngāi Tahu Claims Settlement Act prohibits the targeted commercial harvest of “Kanakana/Ute – southern lamprey (*Geotria australis*)”. The Ngāti Ruanui and Ngāti Mutunga Treaty settlements specifically prohibit the commercial harvest of lamprey within their protocol areas unless the Minister can demonstrate a commercial harvest is sustainable.

Concern over the state of the customary fishery has led some mana whenua to initiate examining customary harvest methods as a way to monitor lamprey abundance (e.g., Te Ao Marama Inc and Waikawa Whanau 2010; Kitson et al. 2012).

5.3 Life cycle, age and growth

Lamprey are anadromous and have a complex life cycle (Figure 5-3). Adult lamprey migrate from the marine environment between April and November to reach spawning habitat in fresh water. They can travel substantial distances inland (up to 200 km and can climb over 300 m in elevation). Their upstream migrations are stimulated by increases in stream discharge and can occur during the day but are most often found at night. Migrations are linked to the receding flood waters and occur on both small and large flood flows. Adult lamprey are attracted to pheromones released by juveniles and it is believed that this helps adult locate suitable spawning and rearing habitats (Baker et al. 2016). Adults are in freshwater for > 1 year before they spawn (Glova 1995). Adults are excellent climbers of obstacles, even vertical surfaces.

Baker et al. (2017) documented lamprey spawning habitat in Aotearoa-New Zealand, where mating pairs spawned egg clusters under large boulders and remained with eggs/larvae post-hatching. At present, it appears that the male has an active role in caring for the developing larvae and assists in hatching. Based on limited observations, the role the female takes during nesting is uncertain. However, both sexes survive spawning for three months, which is the longest documented post-spawning survival of any lamprey species worldwide. Spawning was followed by adult mortality, up to 105 days later. Larval lamprey can remain in the nest for at least two weeks before dispersal, as opposed to dispersing immediately post hatching (Baker et al. 2017).

The juveniles or ammocoetes are benthic worm-like filter feeding organisms (~100 mm in length) that prefer sites with sandy substrates and high detrital loads (Potter et al. 1986; Jellyman and Glova 2002a). Ammocoetes remain in fresh water until metamorphosis (3.25 to 4.25 years; Potter and Hilliard 1986), after which the sub-adult (macrophthalmia) migrates downstream to marine adult habitats. Here, they attach onto the body of other fish and to marine mammals and live as parasites. Length of the marine stage is not precisely known but it is suggested to range between two and four years (McDowall 2000; Salas et al. 2015).

There are few estimates of lamprey demographic parameters in Aotearoa-New Zealand. The mean body size of mature *G. australis* is 50–60 cm total length. Ammocoetes can grow 0.10 mm day⁻¹ and 0.00038 g day⁻¹ when held in the laboratory and fed ad libitum (Kelso 1993). Key demographic parameters are shown in Table 5-2.

Table 5-2: Key demographic characteristics for pouched lamprey, *G. australis*.

Key demographic parameters	Estimates	Reference
Maximum size (mm)	620	Fernholm (1990) ³
Minimum size sexual maturity (mm)	–	–
Age at sexual maturity (yrs)	–	–
Longevity (yrs)	–	–
Growth parameter (K)	–	–
Fecundity	48,004 to 68,212 eggs	Renaud (2011); Baker et al. (2017)
Larval size _{hatch} (mm)	7.2 (total length) 0.30 (standard error)	Baker et al. (2017)
Egg size (mean; mm)	1.18	Baker et al. (2017)
Generation length (yrs)	9 years	Closs et al. (2014) ¹

5.4 Distribution and environmental preferences

Lamprey are widely distributed across the temperate zone of the southern hemisphere, except for South Africa. They can be found in south-western and south-eastern Australia, Aotearoa-New Zealand and southern South America (McDowall 2000; Allen et al. 2002; James 2008). Although electrophoretic analysis has previously indicated genetic homogeneity of stocks in Australia and Aotearoa-New Zealand (Johnston et al. 1987), Neira et al. (1988) found some obvious morphological differences among the ammocoetes of *G. australis* from Australasia (Australia and Aotearoa-New Zealand), Argentina, and Chile. It is now believed that further *Geotria* comprises more than a single species (Baker per comm. 2019).

Data on lamprey distribution are limited because no monitoring programs exist in Aotearoa-New Zealand and lamprey are not usually the target species in fish surveys (Bice et al. 2019). The low numbers of observations recorded in the NZFFD are likely to be due to the difficulties in capturing this species. Juveniles occupy the substrates of riverbeds and are difficult to capture with an electric-fishing machine. Adults migrating back from the sea to spawn also bury themselves in river substrates and are difficult to capture. Thus, lamprey tend to be underrepresented in standard fish surveys so their distribution may be more extensive than the NZFFD suggests. Little is known about the distribution of the marine phase, but individuals may travel large distances. According to Leathwick et al. (2008) distribution models, segment summer temperatures is the environmental variable associated with their distribution (Table A-1).

Lamprey are broadly distributed throughout the North and South Islands, as well as the Chatham Islands and Stewart Island (James 2008; Figure 5-4). Lamprey are relatively common in a few areas such as Taranaki, Wellington, Banks Peninsula, and the south-eastern corner of the South Island, but little known in Northland, Hawkes Bay, and Poverty Bay (Figure 5-4). Lamprey are a taonga species and

³ <https://www.fishbase.se/summary/Geotria-australis>

Southland is considered a nationally important area for this species with significant populations found in the Mataura and Waikawa Rivers.



Figure 5-4: Lamprey (*G. australis*) distribution in Aotearoa-New Zealand. Locations of NZFFD records where kōaro are present (black circles) and absent (grey circles). Source: NIWA.

5.5 Diet and predation

During marine life, lamprey feed parasitically on fish and potentially marine mammals (Hilliard et al. 1985; Renaud et al. 2009) but the species in question are not fully known. It is possible that trout

predation on juveniles is a threat to lamprey populations (B. David, pers. obs. in Closs et al. 2014). There are also accounts of lamprey being preyed upon by tuna and seals (J. Kitson, pers. comm.).

5.6 Recruitment

Relationships between numbers of reproductive adults, egg productions and larval survival and the numbers of sub-adults that return to fresh water is unknown. Lamprey recruitment is not well described, and they are rarely caught upon entry to fresh water.

5.7 Current impacts of climate change

Impacts of climate change are largely unknown for this species. In Australia, species distribution models developed by Bond et al. (2011) linked the probability of occurrence of pouched lamprey with physiographic, climatic, and hydrologic characteristics of Victorian rivers. The distribution models were then used to derive probabilities of pouched lamprey occurrence throughout the stream network based on baseline (historic) and future climate scenarios (Bond et al. 2011).

Climate scenarios corresponded to the A1T (low), A2 (median) and A1FI (high) scenarios from the Intergovernmental Panel on Climate Change (IPCC)'s Special Report on Emission Scenarios (SRES marker scenarios IPCC 2000; see Bond et al. 2011), with associated temperature increases (by 2030) of +0.54 °C, +0.85 °C and +1.24 °C, respectively. Results indicated more than 30% reduction in pouched lamprey distributions at the lower (+0.54 °C), median (+0.85 °C) and upper (+1.24 °C) climate change scenarios. Data does not exist for Aotearoa-New Zealand populations.

Climate change and drought may reduce the quality of juvenile habitat (increased temperature, reduced dissolved oxygen), particularly in streams with altered flow regimes, and migratory cues and connectivity for freshwater entry (Bice et al. 2019). In Australia, lamprey are predicted to lose between 50% and 70% of their current population size as a result of temperature increases (Stewart et al. 2018).

5.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between lamprey and the wider ecosystem include:

- Lamprey require a host species during the marine life phase to complete their life cycle;
- Given their diadromous life cycle, lamprey play an important role in marine-freshwater nutrient and subsidy transfer;
- Lamprey can be caught as bycatch in whitebait and commercial eel nets (J. Kitson, pers. comm.).

5.9 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). The stressors derived from the IUCN threat assessment (n=2), included: 'dams and water management/use [large dams]'; and 'agriculture and aquaculture [small-holder grazing, ranching or farming]' (see Supplementary Appendix 2). In addition, more detailed information about stressors on lamprey populations are summarised from Bice et al. (2019) as:

- River regulation and construction of barriers (barrages, weirs and dams) have likely affected the abundance and distribution of the species. The species has strong climbing ability, but large

dams impede passage. Hydropower development is a specific threat in Aotearoa-New Zealand and Chile. In cases where upstream migration is completely obstructed, spawning and recruitment may occur if appropriate spawning and nursery habitats exists downstream of barriers. Nonetheless, the location of spawning and nursery sites is unknown for many of the rivers inhabited by pouched lamprey. Whilst measures are being implemented to mitigate barriers (e.g., fishway construction), obstruction of migration will remain an issue into the future. Trap and transfers from below dams to areas upstream may benefit this species (Closs et al. 2014; Baker et al. 2016). Although lamprey are able to climb short vertical surfaces, poorly designed instream barriers like culverts, weirs and fords can impact the upstream migration of adult fish;

- Changes to river flow regimes (i.e., reductions in overall discharge and altered seasonality) have likely altered migratory cues (presence of juvenile pheromones) and the quality of juvenile nursery habitats. As of 2012, reductions in 'free-flowing' reaches of rivers across the species range, as a result of dam construction, ranged 0–50% (Liermann et al. 2012). In addition, altered land use (conversion to farmland) has likely resulted in degradation of spawning/nursery habitats (e.g., siltation) in many parts of the species range. In Aotearoa-New Zealand, adults now likely have to travel greater distances to find suitable spawning habitat (as a result of land use), which may impact on adult condition, whilst the overall area of suitable spawning habitat has likely decreased;
- The boulder habitat utilised for spawning and nesting is expected to have been reduced nationwide through conversion of most of forest to farmland (Closs et al. 2014) and the installation of hydroelectric dams;
- Lamprey Reddening Syndrome (LRS) was observed in spring 2011 (Mataura River, Southland) and mass mortality occurred of pre-reproductive adults. The cause of this syndrome is unknown, but LRS may pose a real threat to declining lamprey populations especially in Southland which is a nationally significant area for this species (Closs et al. 2014). LRS has also been observed in Otago, Canterbury and Taranaki regions (Brosnahan et al. 2019).

5.10 Critical data gaps and level of uncertainty

Pouched lamprey are difficult to sample due to their cryptic habits. Few targeted monitoring programs exist for the species across its broad range, and as such data on distribution and population size are limited. However, there have been population declines in Aotearoa-New Zealand and Australia. Records are largely limited to incidental captures in broader fish monitoring programs. Due to the lack of data and specific information currently available from across this species' range, particularly on the distribution, population trends and impacts of threats to this species, pouched lamprey are assessed as 'Data Deficient' according to the IUCN threat classification system (Bice et al. 2019).

5.11 References

- Allen, G.R., Midgley, S.H., Allen, M. (2002) Field guide to the Freshwater Fishes of Australia. Western Australian Museum, Perth.
- Baker C.F., Stewart M., Reeve, K. (2016) Lamprey pheromones in the Clutha/Mata-Au catchment. *NIWA Client Report HAM2016-068*. 20 p.

- Baker, C.J., Jellyman, D.F., Reeve, K., Crow, S., Stewart, M., Buchinger, T., Li, W. (2017) First observations of spawning nests in the pouched lamprey (*Geotria australis*). *Canadian Journal of Fisheries and Aquatic Sciences* 74: 1603–1611.
- Bice, C., Gorski, K., Closs, G., Franklin, P., David, B., West, D., Crow, S., Allibone, R., Ling, N., Hitchmough, R. (2019) *Geotria australis*. The IUCN Red List of Threatened Species 2019: e.T197275A123377152. <https://dx.doi.org/10.2305/IUCN.UK.2019-3.RLTS.T197275A123377152.en>. Downloaded on 28 January 2020.
- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions*, 20: 931–942.
- Brosnahan, C.L., Pande, A., Keeling, S.E., van Andel, M., Jones, J.B. (2019) Lamprey (*Geotria australis*; Agnatha) reddening syndrome in Southland rivers, New Zealand 2011–2013: laboratory findings and epidemiology, including the incidental detection of an atypical *Aeromonas salmonicida*. *New Zealand Journal of Marine and Freshwater Research*, 53: 416–436, DOI: 10.1080/00288330.2018.1556167
- Closs, G., Franklin, P., David, B., West, D., Crow, S., Allibone, R., Ling, N., Hitchmough, R. (2014) *Geotria australis*. The IUCN Red List of Threatened Species 2014: e.T197275A2480788. <http://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T197275A2480788.en>. Downloaded on 05 July 2017.
- Fernholm, B. (1990) Petromyzontidae. p. 79-80. In O. Gon and P.C. Heemstra (eds.) *Fishes of the Southern Ocean*. J.L.B. Smith Institute of Ichthyology, Grahamstown, South Africa: 462.
- Glova G. (1995) The secret life of the lamprey. *Water & Atmosphere*, 3:20–21.
- Hilliard, R.W., Potter, I.C., Macey, D.J. (1985) The Dentition and Feeding Mechanism in Adults of the Southern Hemisphere Lamprey *Geotria australis* Gray. *Acta Zoologica*, 66: 159–170.
- IUCN. (2019) The IUCN Red List of Threatened Species. Version 2019-3. Available at: www.iucnredlist.org. (Accessed: 10 December 2019).
- James, A. (2008) Ecology of the New Zealand Lamprey (*Geotria australis*). A literature review. Department of Conservation, Whanganui, New Zealand: 28.
- Jellyman, D.J., Glova, G.J. (2002a) Habitat use by juvenile lampreys (*Geotria australis*) in a large New Zealand river. *New Zealand Journal of Marine and Freshwater Research*, 36: 503–510.
- Jellyman, D.J., Glova, G.J., Sykes, J.R.E. (2002b) Movements and habitats of adult lamprey (*Geotria australis*) in two New Zealand waterways, *New Zealand Journal of Marine and Freshwater Research*, 36: 53–65. DOI: 10.1080/00288330.2002.9517070
- Johnston, P.G., Potter, I.C., Robinson, E.S. Electrophoretic analysis of populations of the southern hemisphere lampreys *Geotria australis* and *Mordacia mordax* *Genetica* 74.2 (1987): 113–117.
- Kelso, J. (1993) Substrate selection by *Geotria australis* ammocoetes in the laboratory. *Ecology of freshwater fish*, 2: 116–120.

- Kitson, J. (2012) Kanakana harvest mātauranga: potential tools to monitor population trends on the waikawa river, southland/murihiku (a scoping project). Contract Number: 10-RF-02: 40.
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- Liermann, C.R., Nilsson, C., Robertson, J., Ng, R.Y. (2012) Implications of Dam Obstruction for Global Freshwater Fish Diversity. *BioScience*, 62: 539–548.
- McDowall, R.M. (1990) New Zealand freshwater fishes: a natural history and guide. Heinemann Reed, Auckland.
- McDowall, R.M. (2000) The Reed guide to New Zealand freshwater fishes. Reed Books, Auckland.
- Neira, F.J., Bradley, J.S., Potter, I.C., Hilliard, R.W. (1988) Morphological variation among widely dispersed larval populations of anadromous southern hemisphere lampreys (Geotriidae and Mordaciidae). *Zoological Journal of the Linnean Society*, (92): 383–408.
- Potter, I.C., Hilliard, R.W. (1986) Growth and average duration of larval life in the southern hemisphere lamprey, *Geotria australis* Gray. *Experientia*, 42: 1170–1173.
- Potter, I.C., Hilliard, R.W., Neira, F.J. (1986) The biology of Australian lampreys. *Limnology in Australia*.
- Renaud, C.B. (2011) Lampreys of the world. An annotated and illustrated catalogue of lamprey species known to date. FAO Species Catalogue for Fishery Purposes. No. 5. Rome, FAO. 109 pp.
- Salas, C.A., Yopa, K.E., Warrington, R.E., Hart, N.S., Potter, I.C., Collin, S.P. (2015) Ontogenetic shifts in brain scaling reflect behavioural changes in the life cycle of the pouched lamprey *Geotria australis*. *Frontiers in Neuroscience*, 9: 251.
- Stewart, B.A., Ford, B.M., Van Helden, B.E. (2018) Incorporating climate change into recovery planning for threatened vertebrate species in southwestern Australia. *Biodiversity Conservation*, 27: 147–165. <https://doi.org/10.1007/s10531-017-1426-5>

6 Kātaha/Aua/Yellow-eye mullet (*Aldrichetta forsteri*)



© NIWA M. Francis

Figure 6-1: Yellow-eye mullet (*Aldrichetta forsteri*). Source: NIWA.

6.1 Habitats

Yellow-eye mullet (Figure 6-1) are found in four habitat types according to the IUCN threat assessment (David et al. 2014; see Supplementary Appendix 2). They are found in four marine neritic habitat types (estuaries; subtidal sandy; subtidal sandy mud and seagrass (submerged)). They are essentially a marine fish but frequent estuaries, harbours and coastal lakes. Yellow-eye mullet can tolerate a range of salinities, but they cannot live permanently in fresh water. They periodically enter tidal areas, but often only for a duration of one or two tidal cycles. The adults are most commonly seen in shoals in shallow waters at depth of up to 10 m (Bray and Gomon 2011).

6.2 The fishery

When managing yellow-eye mullet, Fisheries New Zealand utilises a suite of management tools including: Total Allowable Catch (TAC) and Total Allowable Commercial Catch (TACC); and on-commercial allowances for customary and recreational fishing. Yellow-eye mullet supports minor commercial fisheries in Aotearoa-New Zealand and is used for human and animal consumption and as fish bait (McDowall 2000; Bray and Gomon 2011; Fisheries New Zealand 2018). The fishery is subdivided into 10 management units (Figure 6-2). The commercial fishery predominantly uses set-nets and gillnets.

In Aotearoa-New Zealand, landings reached a TACC peak of 68 tonnes in 1986–87. This species entered the Quota Management System (QMS) on 1 October 1998 and total landings in 1998–99 were 34 tonnes. Catches declined to a minimum of 13 tonnes in 2004–05. Landings subsequently increased, with peaks of 38 tonnes in 2010–11 and 45 tonnes in 2014–15. Since this fishery was introduced to the QMS TACCs have remained below the 68-tonne limit (Figure 6-3; Fisheries New Zealand 2018). The total asset value of the yellow-eye mullet commercial fish resource caught under the QMS (year ended September 2018) was NZ\$300,000⁴.

Strong seasonal trends are evident in the catch data, with annual peaks mostly in July-August, indicating a winter fishery in Aotearoa-New Zealand. Yellow-eye mullet is a popular recreational species (McDowall 2000; Bray and Gomon 2011; Fisheries New Zealand 2018) where the commercial, recreational and customary fishery allowances are 68, 52 and 36 tonnes respectively. This species is also an important commercial and recreational species in new South Wales, South Australia, Tasmania and Western Australia (Earl and Ferguson 2013).

⁴ <https://figure.nz/chart/LM7iOMLt6SEQscZR>

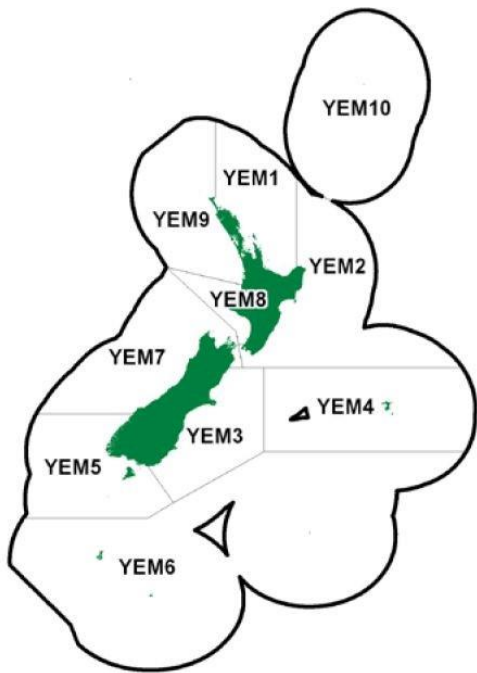


Figure 6-2: Yellow-eye mullet fishery management units. YEM = Yellow-eye mullet. Source: MPI (2008).

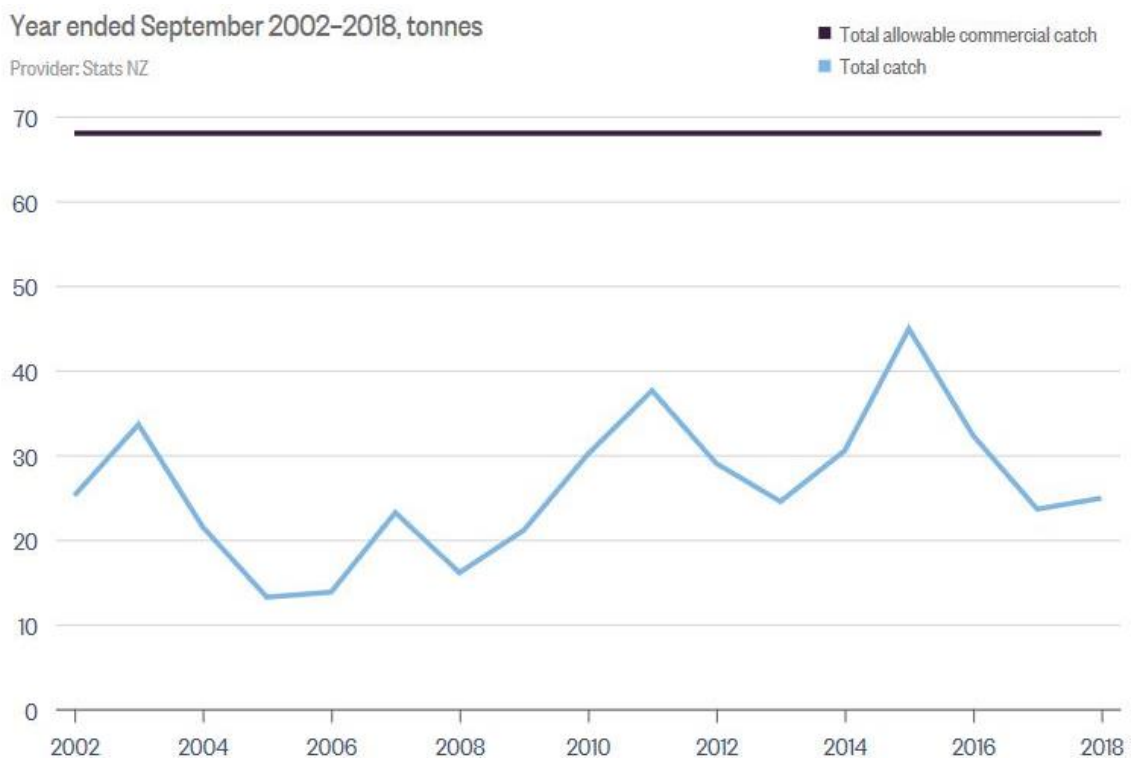


Figure 6-3: Total commercial catch (blue line) of yellow-eye mullet in Aotearoa-New Zealand between 2002–03 and 2017–18. Since this fishery was introduced to the QMS TACCs have remained below the 68 tonne limit (black line). Source: Figure.NZ and Stats NZ (<https://figure.nz/chart/XuZ4uvsOpxD1Tz8E-bNEAoFUDUChDThQa>).

One of the most significant fisheries is Te Waihora/Lake Ellesmere. The commercial fishery in Te Waihora fluctuates with both recruitment of fish from the sea, and also with market demands (Jellyman 2012). Crow and Bonnett (2013) summarised the literature on yellow-eye mullet in this system and their key findings are reproduced here:

- In an early review of the mullet fishery of the lake, Gorman (1960) noted that the catches peaked in midwinter with a lesser peak in summer. While this winter fishery was partly because eels were less active (and eels frequently attacked mullet caught in the gill nets), it was also because mullet are affected by cold water temperatures which results in them forming schools and becoming more accessible to fishers (Jellyman and Smith 2008);
- Catches of mullet in Te Waihora and at other locations around Canterbury show marked seasonal variability (Jellyman 2012). Fyke nets set at Taumutu (Te Waihora) gave peak catches from May to July and in December (Hardy 1989), but few mullet were caught at Timberyard Point during this time. In contrast, seasonal seine net catches from September 2005 to May 2008 in Te Waihora captured no mullet from 60,000 fishes (NIWA unpublished data);
- Low proportions (0.05% of catch abundance) of mullet were caught in a study of fish recruitment by Taylor and Graynoth (1995) with most fish being caught during July, September and October. Similarly, 66% of the mullet recorded by Webb (1972) in the Avon/Heathcote estuary were caught between September and December.

In a seasonal study of fish entering the Rakaia Lagoon, Eldon and Greager (1980) found that mullet were present from September to June, but large numbers from only January to April. In a seasonal study of the Waimakariri Lagoon, Eldon and Kelly (1985) recorded juvenile mullet (less than 60 mm) during February and March. As movement into brackish and fresh water is not obligatory for this species, it is quite likely that recruitment of larger fish could occur at almost any month of the year.

6.3 Life cycle, age and growth

Yellow-eye mullet form large aggregations prior to spawning. Spawning occurs at sea and there is some evidence that biennial spawning occurs, with peaks in winter and summer. Yellow-eyed mullet appear to leave their estuarine habitat to spawn in coastal waters producing several hundred thousand eggs, which float in the surface waters of the sea up to 33 km offshore. Egg development begins in July and maturity occurs by late December. For both sexes, first maturity is reached at 220-230 mm. This species commonly reaches a total length of around 300 mm, but it has been reported up to 500 mm and a maximum age of 7 years (Table 6-1) (Curtis and Shima 2005; McDowall 2000; MPI 2008; Bray and Gomon 2011; Earl and Ferguson 2013).

Table 6-1: Key demographic characteristics for yellow-eye mullet (*Aldrichetta forsteri*).

Key demographic parameters	Estimates	Reference
Maximum size (mm)	500	McDowall (1990)
Minimum size sexual maturity (mm)	223 (Female); 229 (Male)	Earl and Ferguson (2008)
Age at sexual maturity (yrs)	2-3	Earl and Ferguson (2008)
Longevity (yrs)	7	Earl and Ferguson (2008)
Growth parameter (K)	0.079	Earl and Ferguson (2008)
Fecundity	–	–
Larval size _{hatch} (mm)	–	–
Egg size (mean)	–	–
Generation length	–	–

6.4 Distribution and environmental preferences

Yellow-eye mullet are widely distributed in temperate waters of southern Australia, from just north of Sydney (New South Wales) to Shark Bay (Western Australia), around Tasmania and throughout the whole of Aotearoa-New Zealand, including the North and South Islands and Chatham Island (McDowall 2000, Bray and Gomon 2011). According to the NZFFD appears their distribution seems patchy with a few records around the country (Figure 6-4), but they are likely found in more locations.

The preferred temperature range of yellow-eye mullet is 13.5–20.2°C, with a mean 17.1 °C⁵. A recent study found that when yellow-eye mullet were raised at different temperatures (between 13°C and 21°C) their metabolic characteristics adjusted to variable temperate environmental temperatures and they managed the energetic costs of digestion and feed assimilation (Flikac et al. 2020). Yellow-eye mullet have a large aerobic scope suggesting physiological adaptation to the highly variable temperate coastal environment (Flikac et al. 2020).

⁵ <https://www.fishbase.se/summary/12938>

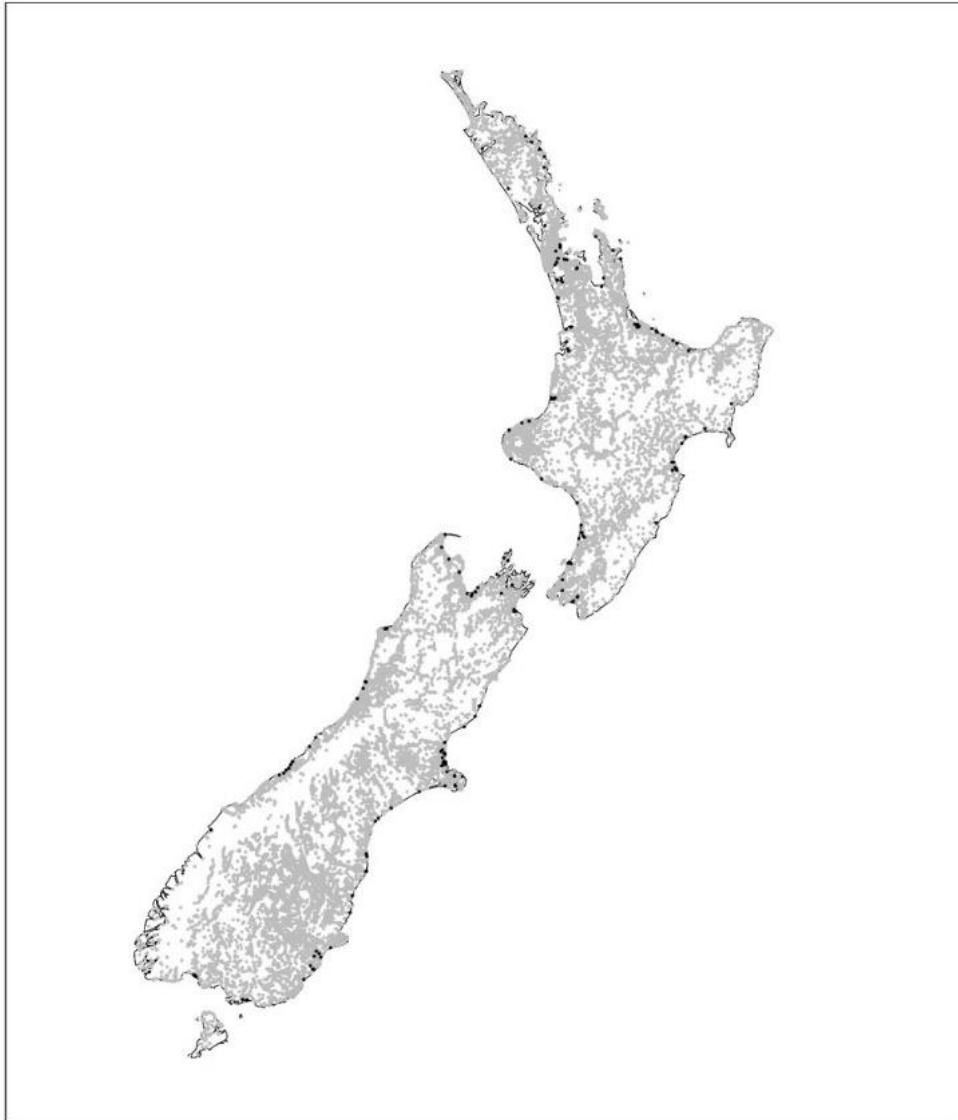


Figure 6-4: Yellow-eye mullet distribution in Aotearoa-New Zealand. Locations of NZFFD records where yellow-eye mullet are present (black circles) and absent (grey circles). Source: NIWA.

6.5 Diet and predation

Yellow-eyed mullet are omnivorous and feed on a wide range of food types including algae, crustaceans, diatoms, molluscs, insect larvae, fish, polychaetes, coelenterates, fish eggs and detritus (David et al. 2014). Yellow-eye mullet are eaten by juvenile fur seals (Page et al. 2005) and seabirds such as Australasian gannets (*Morus serrator*) and cormorants (Robertson 1992).

6.6 Recruitment

Recruitment information is limited for this species and there is no information on relationships between spawning stock biomass, egg production and survival. There is no information on the age of recruitment into estuarine waters. Within estuaries and river systems yellow-eyed mullet are separated to some extent by age, with older fish preferring more saline water and juveniles sometimes found in fresh water.

6.7 Current impacts of climate change

Impacts of climate change are largely unknown for this species. In Australia, species distribution models were developed by Bond et al. (2011) that linked the probability of occurrence of yellow-eye mullet with physiographic, climatic, and hydrologic characteristics of Victorian rivers. The distribution models were then used to derive probabilities of yellow-eye mullet occurrence throughout the stream network based on baseline (historic) and future climate scenarios (Bond et al. 2011).

Climate scenarios corresponded to the A1T (low), A2 (median) and A1FI (high) scenarios from the Intergovernmental Panel on Climate Change (IPCC)'s Special Report on Emission Scenarios (SRES marker scenarios IPCC, 2000; see Bond et al. 2011), with associated temperature increases (by 2030) of +0.54 °C, +0.85 °C and +1.24 °C, respectively. Results indicated >30% increase in yellow-eye mullet distributions at the lower (+0.54 °C), median (+0.85 °C) and upper (+1.24 °C) climate change scenarios. Data does not exist for Aotearoa-New Zealand populations.

Finfish recruitment surveys along the west and south coasts of Australia during 2011 and 2012 indicated a marked decline of the “cool temperate” yellow-eye mullet on both coasts. These results are being investigated with respect to the 2011–2012 marine heatwave and are continually monitored (Caputi et al. 2014).

Recent mortalities of yellow-eye mullet have been observed in Hawkes Bay. Low river levels and water temperatures close to 29°C resulted in increased infections in fish and eventually mortalities⁶.



Figure 6-5: Dead yellow-eye mullet in the Esk River Hawkes Bay 2020. Source: Hawkes Bay Regional Council.

6.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between yellow-eye mullet and the wider ecosystem include:

⁶ https://www.nzherald.co.nz/hawkes-bay-today/news/article.cfm?c_id=1503462&objectid=12308061

- As a predator and prey species;
- Nutrient subsidy to fresh waters;
- Potential bycatch of seabirds, dolphins and other marine mammals in the set net fishery location dependent. Estimated catch of 75 seabirds annually⁷.

6.9 Additional (multiple) stressors

The stressor derived from the IUCN threat assessment (n=1) were 'fishing and harvesting of aquatic resources [subsistence, small scale harvest]' (see Supplementary Appendix 2). Other stressors include:

- The highly localised nature of the fishery for this species and the relatively high landings taken in specific areas (e.g., the Manukau Harbour in Aotearoa-New Zealand), may make this species susceptible to localised depletion. However, as this species is widespread and common throughout its range, its global population is considered stable and its threat status has been assessed as being of 'Least Concern' (David et al. 2014);
- Sedimentation of nearshore and estuaries can compromise the productivity of the yellow-eye mullet fishery because of alterations to their habitats. These include the modification or loss of important fish nursery habitats (i.e., sea-grass meadows) that may be vital nursery grounds for this species⁸;
- High nutrient levels can cause 'blooms' of phytoplankton and algae. This can significantly alter predator/prey relationships in an area; it can reduce light levels (affecting species like seagrass and the fish that rely on this habitat); and can also lead to eutrophication of the seabed. Particularly, this affects habitats in sheltered waters and can potentially affect fisheries that rely on habitats (i.e., yellow-eye mullet) in these places for some part of their life cycle⁹.

6.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- The biological stock structure for yellow-eye mullet is uncertain and further studies are required to confidently define biological stock delineation for this species. Ageing of fish stocks is required in Aotearoa-New Zealand;
- Estimates of current and reference biomass are not available. It is not known if recent catch levels are sustainable or at levels that will allow the stock to move towards a size which will support the Maximum Sustainable Yield (MPI 2008);
- The extent of marine-freshwater movements is largely unknown. Little is known about their spawning and juvenile habitat requirements.

⁷ <http://bestfishguide.org.nz/seafood/yellow-eyed-mullet/#Bycatch-threatened>

⁸ <https://fs.fish.govt.nz/Page.aspx?pk=116&dk=579>

⁹ <https://fs.fish.govt.nz/Page.aspx?pk=116&dk=579>

6.11 References

- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions*, 20: 931–942.
- Brad, P., Jane, M., Simon, D.G. (2005) Dietary resource partitioning among sympatric New Zealand and Australian fur seals. *Marine Ecology Progress Series*, 293: 283–302.
- Bray, D.J., Gomon, M.F. (2011) *Aldrichetta forsteri* in Fishes of Australia, accessed 31 Mar 2020, <http://136.154.202.208/home/species/4534>
- Caputi, N., Jackson, G., Pearce, A. (2014) The marine heat wave off Western Australia during the summer of 2010/11 – 2 years on. Fisheries Research Report No. 250. Department of Fisheries, Western Australia: 40.
- Cook, D.G., Jaksons, P., Jerrett, A.R. (2018) Tagging investigations with small estuarine-associated fish: tag evaluation, capture methodologies and assessment of capture stress and survival in yellow-eyed mullet *Aldrichetta forsteri*. *Marine and Freshwater Research*, 69: 1595–1603.
- Crow, S., Bonnett, M. (2013) Te Waihora Mahinga Kai: a compilation of data and summary of existing research on freshwater fishes in Te Waihora. Whakaora Te Waihora Partners.
- Curtis, T.D., Shima, J.S. (2005) Geographic and sex-specific variation in growth of yelloweyed mullet, *Aldrichetta forsteri*, from estuaries around New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 39(6): 1277–1285.
- David, B., Franklin, P., Closs, G., Hitchmough, R., Ling, N., Allibone, R, Crow, S., West, D. (2014) *Aldrichetta forsteri* (errata version published in 2018). The IUCN Red List of Threatened Species 2014: e.T197036A135873977. <https://dx.doi.org/10.2305/IUCN.UK.2014-3.RLTS.T197036A2478220.en>. Downloaded on 17 February 2020.
- Earl, J., Ferguson, G.J. (2013) Yelloweye mullet (*Aldrichetta forsteri*) stock assessment report 2011–12. Report to Primary Industries and Regions South Australia (Fisheries and Aquaculture). SARDI Publication F2007/001048-1. SARDI Research Report Series 737. South Australian Research and Development Institute (Aquatic Sciences), Adelaide.
- Eldon, G.A., Greager, A.J. (1983) Fishes of the Rakaia lagoon. Fisheries Environmental Report. Christchurch, New Zealand Ministry of Agriculture and Fisheries 30. 65 p.
- Eldon, G.A., Kelly, G.R. (1985) Fishes of the Waimakariri estuary. Christchurch, New Zealand Ministry of Agriculture and Fisheries. 34 p
- Fisheries New Zealand (2018) Fisheries Assessment Plenary. Stock Assessment and Stock Status Volume 3: Pipi to Yellow-eyed Mullet. May 2018: 606. <https://www.mpi.govt.nz/dmsdocument/18337-fisheries-assessment-plenary-may-2017-volume-3-pipi-to-yellow-eyed-mullet>
- Flikac, T., Cook, D.G., Davison, W.J. (2020) The effect of temperature and meal size on the aerobic scope and specific dynamic action of two temperate New Zealand finfish *Chrysophrys auratus* and *Aldrichetta forsteri*. *Journal of Comparative Physiology B*, 190: 169–183. <https://doi.org/10.1007/s00360-020-01258-5>

- Hardy, C.J. (1989) Fish habitats, fish, and fisheries of the Ellesmere catchment. *New Zealand Freshwater Fisheries Report*, 104. 152 p.
- Jellyman, D.J., Smith, C. (2008) Native Fish and Fisheries of Te Waihora. In: K.F.D. Hughey K.J.W. Taylor, (Eds.). *Te Waihora/Lake Ellesmere – the 2007 State of the Lake and Management Futures Report*. 41-48.
- Jellyman, D.J. (2012) Fish recruitment into Te Waihora/Lake Ellesmere. A consideration of the requirements of key species. *NIWA Client Report CHC2011- 094*. 75 p
- McDowall, R.M. (1990) *New Zealand freshwater fishes: a natural history and guide*. Heinemann Reed, Auckland.
- McDowall, R.M. (2000) *The Reed guide to New Zealand freshwater fishes*. Reed Books, Auckland.
- MPI (2008) Plenary: Fisheries Assessment Plenary - Yellow-eyed mullet. May 2008: 985-990. https://fs.fish.govt.nz/Doc/5680/YEM_FINAL%2008.pdf.ashx
- Page, B., McKenzie, J., Simon, D.G. (2005) Dietary resource partitioning among sympatric New Zealand and Australian fur seals. *Marine Ecology Progress Series*, 293: 283-302.
- Robertson, D.A. (1992) Diet of the Australasian gannet *Morus Serrator* (G.R. GRAY) around New Zealand. *New Zealand Journal of Ecology*, 16: 77–81.
- Taylor, M.T., Graynoth, E. (1995) Native fish immigration into Lake Ellesmere during 1994. *NIWA Science and Technology Report*, 27: 25.
- Webb, B.F. (1972) Fish populations of the Avon-Heathcote estuary. General ecology; distribution, and length-frequency. *New Zealand Journal of Marine and Freshwater Research*, 6: 570 - 601.

7 Tuna / Shortfin eel (*Anguilla australis*)



Figure 7-1: Adult shortfin eel (*Anguilla australis*). Source: NIWA.

7.1 Habitats

Shortfin eels (Figure 7-1) occupy a diversity of habitat types (n = 10) over their life cycle (see Supplementary Appendix 2). In the marine environment, they are found in pelagic, epi-pelagic and meso-pelagic habitats (Figure 7-2). Juveniles (glass eels [Figure 7-3]) are found in coastal waters, estuaries and the lower reaches of rivers. Sub-adults (i.e., elvers) and adult eels can occupy coastal brackish lakes, intermittent lakes and rivers (see Supplementary Appendix 2).

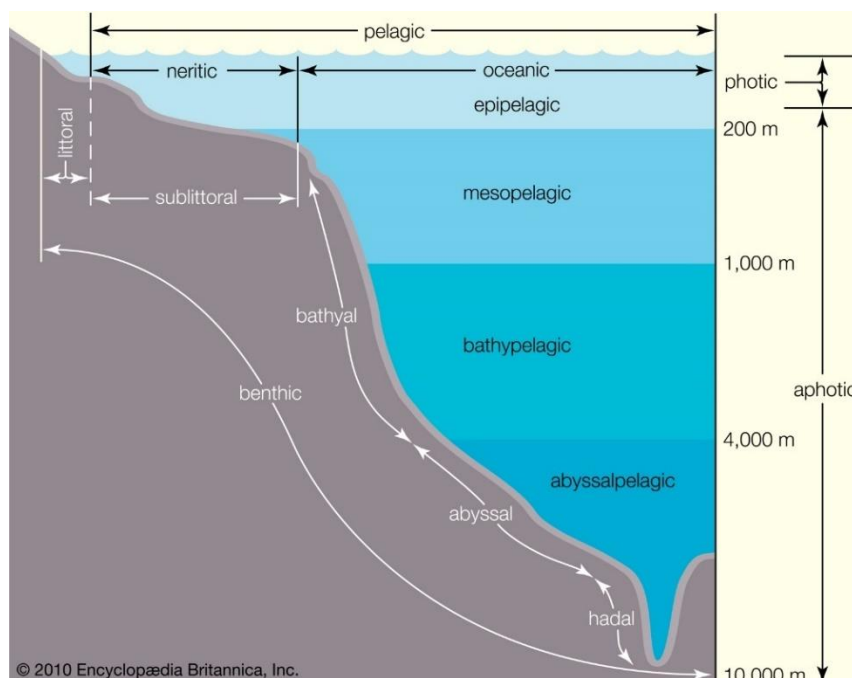


Figure 7-2: Habitat types in the ocean showing the pelagic, epipelagic and mesopelagic habitat types that shortfin eels can be found in. Source: <https://www.britannica.com/science/abyssalpelagic-zone>.

In fresh water, shortfin eels are principally found in lowland habitats: they dominate populations in lowland lakes, estuaries and the lower reaches of rivers. Their upper elevation limit is 700 m (Pike et al. 2019). In shallow lakes such as Te Waihora/Lake Ellesmere (Canterbury) and Lake Waahi (Waikato), juvenile shortfin eels typically prefer gravel and/or muddy substrates in the littoral margins and can be found at depths ranging 0.6–1.2 m (Jellyman and Chisnall 1999). In shallow lakes, larger eels (more than 300 mm) typically prefer sandy substrates, but use a wider range of depths and can be found throughout the lakes (Jellyman and Chisnall 1999). Once shortfins find a suitable habitat, movement may be limited, with an average home range of 30 m (Jellyman and Sykes 2003). Jellyman et al. (1996) reported a study from December 1977 and February 1978 which tagged 4,968 and 4,987 eels respectively and released all fish at LII Bay and the centre of Te Waihora. From these, there were 1,982 recaptures (excluding 780 multiple recaptures where the same eels were caught on more than one occasion) recorded by commercial fishers over the following five years. Shortfin eels were mainly recaptured close to their release site, but some fish did move extensively. A short-term sonic tagging study also found that movement of eels was limited (Jellyman et al. 1996), particularly in areas around the Te Waihora lake margins. Jellyman et al. (1996) reported that while eels tagged in the centre of the lake ranged from 3.3 to 10.8 km from the original tagging location, eels tagged at inshore sites generally remained within 1–2 km of the release site.



Figure 7-3: A group of shortfin glass eels at entry to fresh water. Source: NIWA.

7.2 The fishery

Information on the commercial freshwater eel fishery was largely derived from the executive summaries in Beentjes (2019) and are reported separately here for the North Island and South Island fisheries. When managing freshwater eels, Fisheries New Zealand utilises a suite of management tools including: Total Allowable Catch (TAC) and Total Allowable Commercial Catch (TACC); non-commercial allowances for customary and recreational fishing; minimum and maximum size restrictions; seasonal restrictions to avoid adult migrants, escapement tubes on commercial fyke nets; recreational daily catch limits; and closing areas to commercial fishing, e.g., national parks, parts of the Whanganui, Mōtu and Mōhaka Rivers (Fisheries New Zealand 2018). These tools are not universally applied across the

country and can vary in response to, for example, different fisheries regulations (e.g., Fisheries - South-East Area Commercial Fishing - Regulations 1986) and Treaty of Waitangi settlement mechanisms, e.g., Waikato-Tainui Fisheries Bylaws (Waikato Raupatu River Trust 2014). A recreational individual daily bag limit of six eels was introduced throughout Aotearoa-New Zealand in October 1994; however, there is no quantitative information on recreational harvest. There is also a significant customary fishery for tuna/eels. Customary non-commercial catch allowances range between 1-34 tonnes (Fisheries New Zealand 2018).

The commercial tuna fishery is currently managed as 11 separate longfin and shortfin stocks (Figure 7-4) with the 2016–17 TACC set at 163 tonnes and 580 tonnes, respectively (Clements and Associates, 2016; MPI 2016). New Zealand’s commercial freshwater eel fishery supplies both a domestic and export market and was estimated to have an export value of NZ\$6.1 million by MFish in 2009 (MFish 2009). The asset value of the shortfin eel commercial fish resource caught under the QMS (year ended September 2018) was NZ\$38,300,000¹⁰.

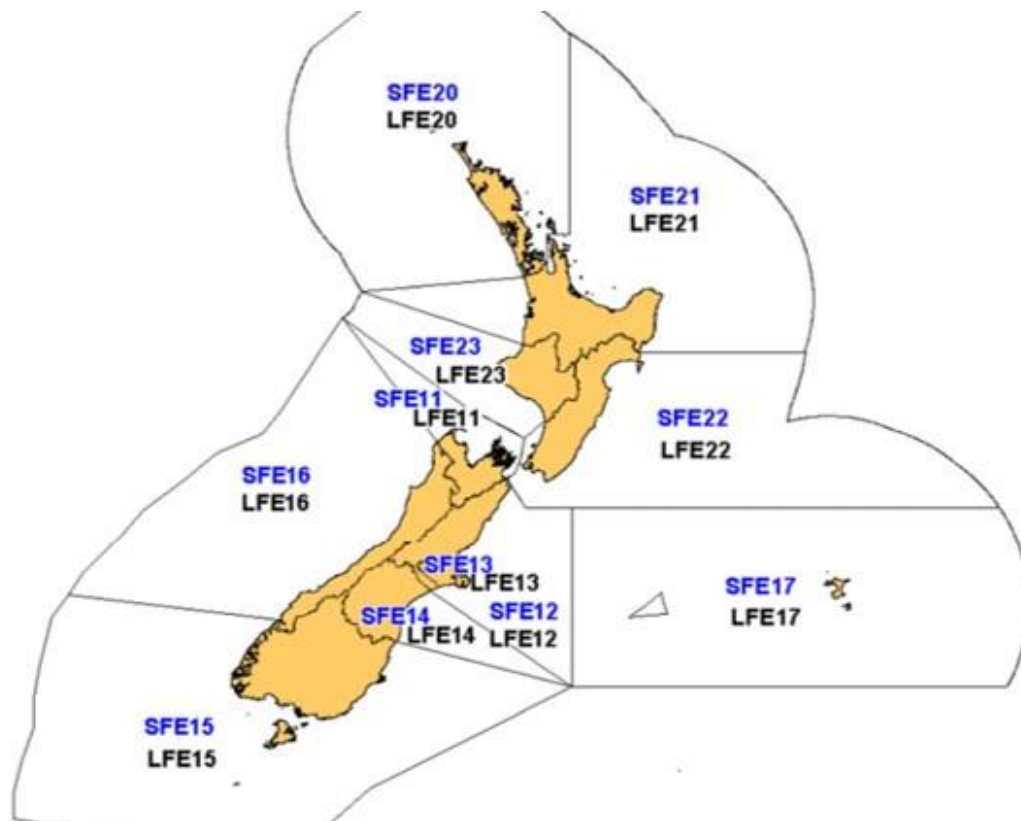


Figure 7-4: Current (as of 2016) quota management areas for shortfin eel (SFE) and longfin eel (LFE).
Source: Fisheries New Zealand (2018).

The shortfin eel fishery was introduced into the QMS in stages between October 2000 to October 2004. Initially South Island longfin and shortfin tuna stocks were combined, and the fishery was managed as 16 fish stocks nationwide. The review of South Island tuna stocks in 2016 led to the separation of the previously combined stocks into individual shortfin and longfin stocks and the subsequent setting of precautionary catch limits and allowances for the separated stocks (Fisheries New Zealand 2018). North Island tuna stocks were last reviewed in 2008, with significant reductions to the TAC (which

¹⁰ <https://figure.nz/chart/LM7iOMLt6SEQscZR>

includes recreational, customary and commercial catch limits) and TACCs of between 10–38% for shortfins and 35–78% for longfins.

The highest shortfin eel total catches since the implementation of the QMS were observed in 2017–18 when 419 tonnes were landed (Figure 7-5).



Figure 7-5: Total commercial catch (blue line) of shortfin eel in Aotearoa-New Zealand between 2002–03 and 2017–18. Source: Figure.NZ and StatsNZ (<https://figure.nz/chart/XuZ4uvsOpxD1Tz8E-Taw2ZuaV2ARStqv/>).

7.2.1 North Island shortfin fishery

The commercial freshwater eel monitoring programme began in the North Island in 2003–04. Between 2003–04 and 2017–18, the North Island shortfin eel fishery was highly aggregated with nearly one-third of the catch originating from just three subareas: AA4 (Dargaville), AD12 (Lake Waikare, Port Waikato), and AC1 (Hauraki plains west). The Dargaville subarea (AA4) contributed 509 t or 12% of the total North Island shortfin catch over this 15-year period and is the most productive subarea in the North Island. There is no consistent trend in the total North Island shortfin annual landed catch over the 15 years, or in the distribution of catch across the three size grades used. The number of subareas for which shortfin catch was landed has declined (from 49 subareas per year, on average for the first six years, to 42 subareas per year on average for the last six years), indicating a slight contraction in the spatial distribution of fishing effort which is consistent with a decline in both the number of permits fished and the fishing events over time. Despite the reduction in effort over time, the landed catch of shortfin has been relatively stable in the key subareas over the 15 years, with no apparent trends (Beentjes 2019).

7.2.2 South Island shortfin fishery

In the South Island, eel processor data were provided initially in 2006–07 for the Quota Management area (QMA) ANG 15 only (Southland/Otago), but no details on specific catch locations were recorded.

Provision of more useful data, similar to the North Island, have been collected for the South Island since the 2010–11 fishing year.

Shortfin in the South Island were caught from most subareas (51 of the possible 58) over the eight years between 2010–11 and 2017–18, but over three-quarters of the catch originated from Te Waihora (AS1 and AS2), and Lake Brunner (AX4). Te Waihora contributed 698 t or 63% of South Island shortfin catch over this eight-year period. There is no consistent trend in annual landed catch, although the proportions of larger eels appear to be declining. There are several factors that might be responsible, including: flood conditions in years that can increase the catch of smaller shortfin, the increase in size grade range in 2017–18, and the split into separate shortfin and longfin stocks in 2016–17 (Beentjes 2019).

The shortfin landed catch has been well below the current shortfin TACC (introduced in 2016–17) as a result of fisher retirements, withheld quota, variable catch contribution from AS2 (Te Waihora migration area), and ACE (Annual Catch Entitlement) imbalances resulting from the nominal 1 t TACCs set in LFE 11 to LFE 14. The pattern of South Island shortfin landed catch by subarea is generally similar over the eight years, except that AS1 and AS2 catches tend to display opposite trends because fishers can catch their quota from either.

7.3 Life cycle, age and growth

Shortfin eels have a catadromous life cycle (Figure 7-6). Most of their life is spent in fresh water, but they migrate to the marine environment for reproduction. Eels have sex-specific life-history strategies. Males may grow faster than females initially, but this difference is soon reversed, and females attain a greater age- and size-at-metamorphosis than males (Davey and Jellyman 2005). The factors determining sex are largely unknown. The sex of developing gonads is labile, and gender is thought to be determined principally by environmental factors. Individuals experiencing rapid growth prior to gonad differentiation tend to develop as males, whereas eels that grow slowly initially are more likely to develop as females (Davey and Jellyman 2005). High eel densities, temperatures and saline conditions are thought to favour development as males, but experimental studies have failed to demonstrate a clear effect on sex determination (Davey and Jellyman 2005). High proportions of female silver eels migrating from some upstream areas, lakes and large rivers may be due to low population densities or poor conditions for growth in these habitats.

Prior to migration, silver eels aggregate in the lower reaches of streams, rivers or at lake outlets waiting for suitable conditions for migration (Jellyman and Unwin 2019). The marine migration of adults to the spawning grounds takes approximately 6–9 months. The migrations of adult eels to the spawning grounds are not well known but their spawning grounds are believed to be somewhere in the westward flowing South Equatorial Current (Jellyman and Bowen 2009; Miller et al. 2009; Figure 7-7). Shortfins from Australia and Aotearoa-New Zealand show small but significant differences in their form/shape (Jellyman 1987; Watanabe et al. 2006), but whether these small differences are a result of spawning in separate areas is unknown. Shortfin eels are semelparous meaning they die after reproduction.

Eggs

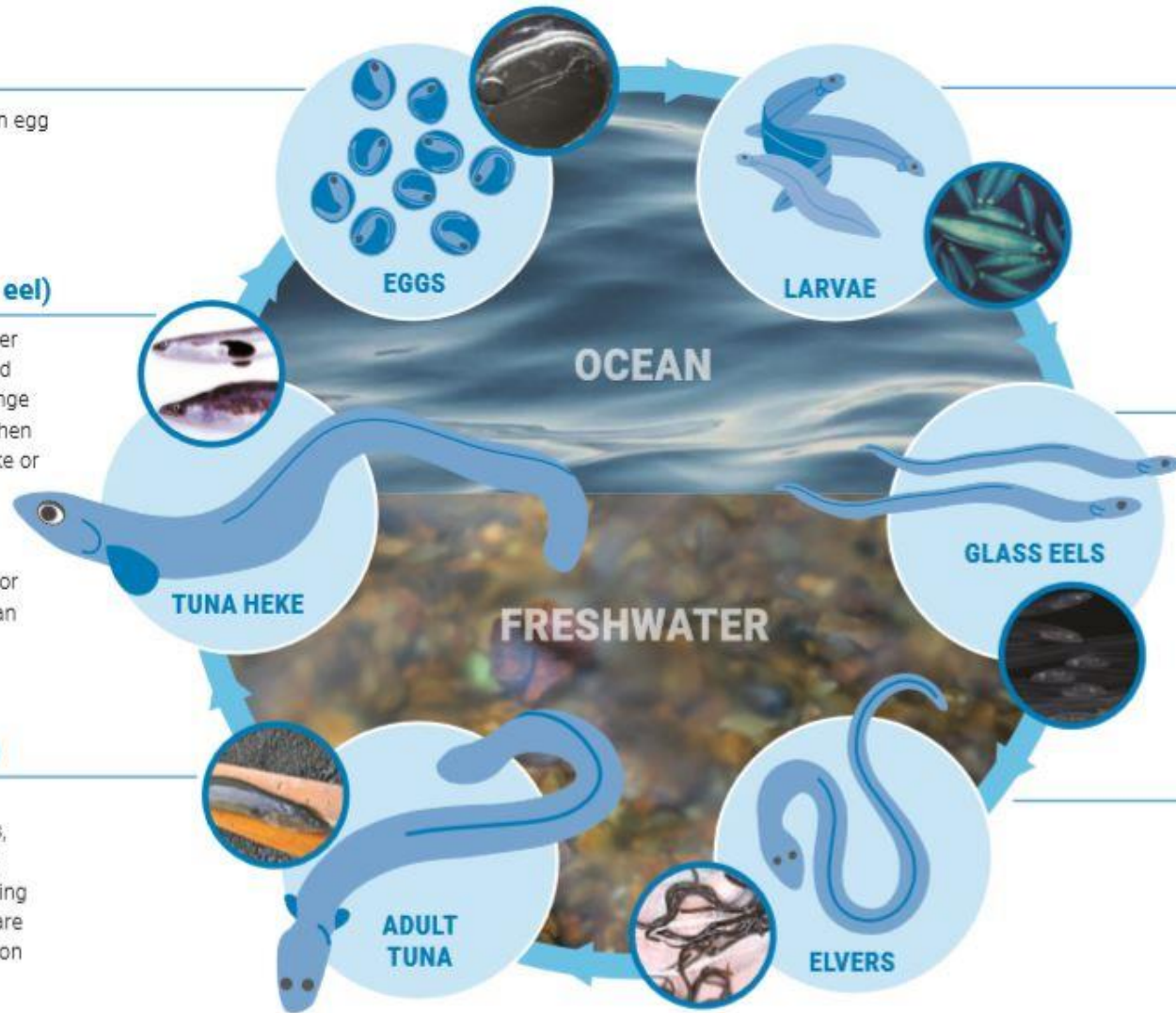
The tuna starts its life as an egg out in the Pacific Ocean.

Tuna Heke (migrant eel)

After a long life in freshwater (on average between 11 and 52 years) tuna start to change and stop feeding. This is when they are known as tuna heke or "silver eels". During rainy nights in autumn (and sometimes spring) they begin their long migration (or journey) to the Pacific Ocean where they spawn and are thought to die.

Adult tuna (feeders)

The adult tuna live for a relatively long time in rivers, lakes, wetlands, ponds and streams, eating and preparing themselves for when they are ready to begin their migration back out to sea.



Larvae

They hatch at sea into see-through (transparent), leaf-shaped, larvae called leptocephalii and spend between 9 to 12 months drifting on ocean currents which bring them back to Aotearoa.

Glass eel

When they reach the seabed near Aotearoa (continental shelf) they change shape and turn into colourless eels called glass eels, about 60-70 mm long. In early spring they move into estuaries, rivers and streams where they rest to get used to their new freshwater environment.

Elvers (juvenile eel)

After several weeks, they begin to turn brown (gain pigmentation) and begin their journey as an elver and head upstream. Keep an eye out in your local waterway during summer for elvers travelling up your stream.

Figure 7-6: Shortfin and longfin tuna life cycle. Source: NIWA.

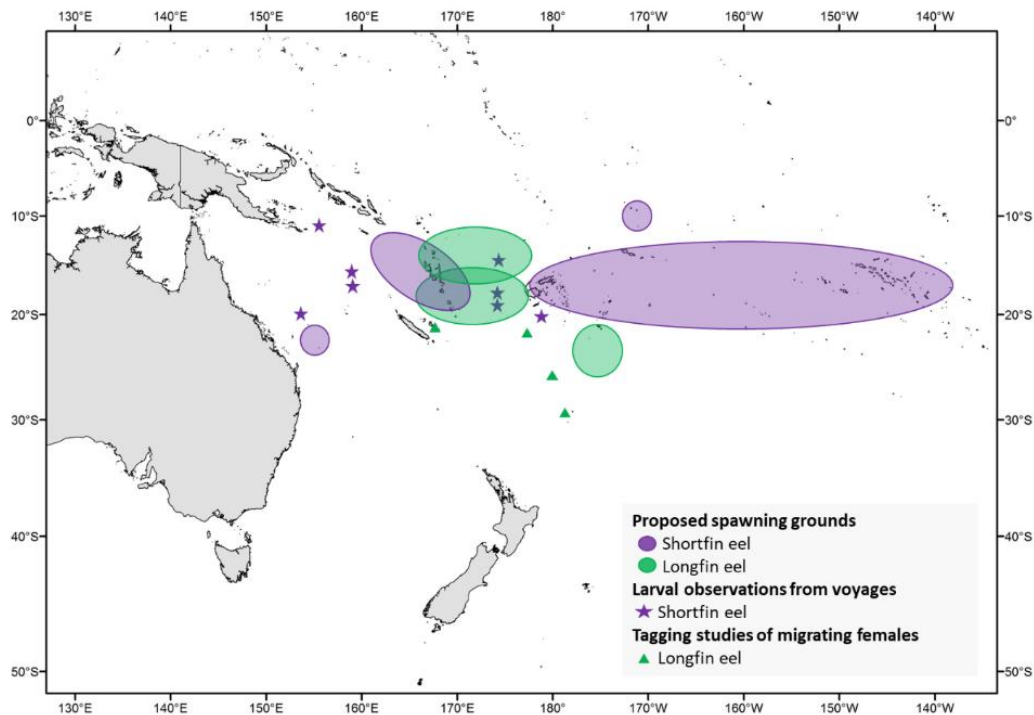


Figure 7-7: Proposed spawning areas of shortfin eel in the Western Pacific Ocean. The purple stars show locations where shortfin eel larvae have been observed. The potential spawning areas of shortfin eel are shown in purple. Source: NIWA.

Following reproduction, shortfin larvae (leptocephalii, Figure 7-5) traverse more than 20° latitude from their potential spawning grounds in the Western Pacific Ocean to Aotearoa-New Zealand (Figure 7-7). Research voyages have found shortfin eel larvae distributed across 10° longitude within the Western Pacific Ocean including east and north-west of Fiji, south of the Solomon Islands, around New Caledonia and in the Coral Sea (Miller et al. 2009; Kuroki et al. 2020). The spatial distribution of shortfin larvae suggests their spawning ground locations may be located either north-east of Samoa, anywhere between Fiji and Tahiti, near west of New Caledonia, or between the north-west of Fiji and west of Vanuatu (Figure 7-8). Shortfin larvae recruiting to Aotearoa-New Zealand are thought to enter the eastward flowing current of the Tasman Front (TF) (Kuroki et al. 2020). Shortfin eel larvae are transported westward and then southward via the East Australian Current (EAC) (Figure 7-7) where some move westward and recruit to eastern Australia. Based on current direction and the similarity in age of leptocephali at metamorphosis, age at capture and the period between metamorphosis and estuarine arrival, Shiao et al. (2001) suggest that Aotearoa-New Zealand glass eels are unlikely to be transported across Tasman Sea from southern Australia by the East Australian Current, and must reach their destination via a different route(s).

Shortfin eels are approximately 7–8 months old when they arrive to Aotearoa-New Zealand’s coastline (Arai et al. 1999; Shiao et al. 2001). The larval stage is on average six months and metamorphosis to the glass eel stage takes approximately one month (Arai et al. 1999). From analysis of otolith microstructure, the spawning times of shortfins spans June to mid-February meaning reproduction occurs in winter, spring and summer (Kuroki et al. 2020). Shortfin glass eels typically enter rivers and streams around Aotearoa-New Zealand between August and December. Regional differences in mean size and condition show an arrival pattern from the north in an anti-clockwise dispersal pattern around Aotearoa-New Zealand.

Shortfin eels reach a maximum size of about 1,100–1,200 mm and 3–4 kg (Jellyman 2013; Williams et al. 2011). The maximum estimated age of shortfin eels is 60 years. The migration of mature adults appears to be dependent on attaining a certain length/weight combination and body condition. Age at spawning migration varies depending on growth rates, with males ranging between 5–22 years, at a size of 400–480 mm, and females 9–41 years and 640–800 mm (Fisheries New Zealand 2018). Unless there is strong evidence of biphasic or asymptotic growth, simple linear growth equations, with a biologically realistic fixed intercept at age 0 (i.e., upon entry to fresh water) or 1, can be used to describe the growth of juvenile and adult eels in Aotearoa-New Zealand (Graynoth and Taylor 2004). Water temperature and eel densities are the most significant drivers of growth rate variation in some rivers (Graynoth and Taylor 2004). Key demographic parameters are shown in Table 7-1.

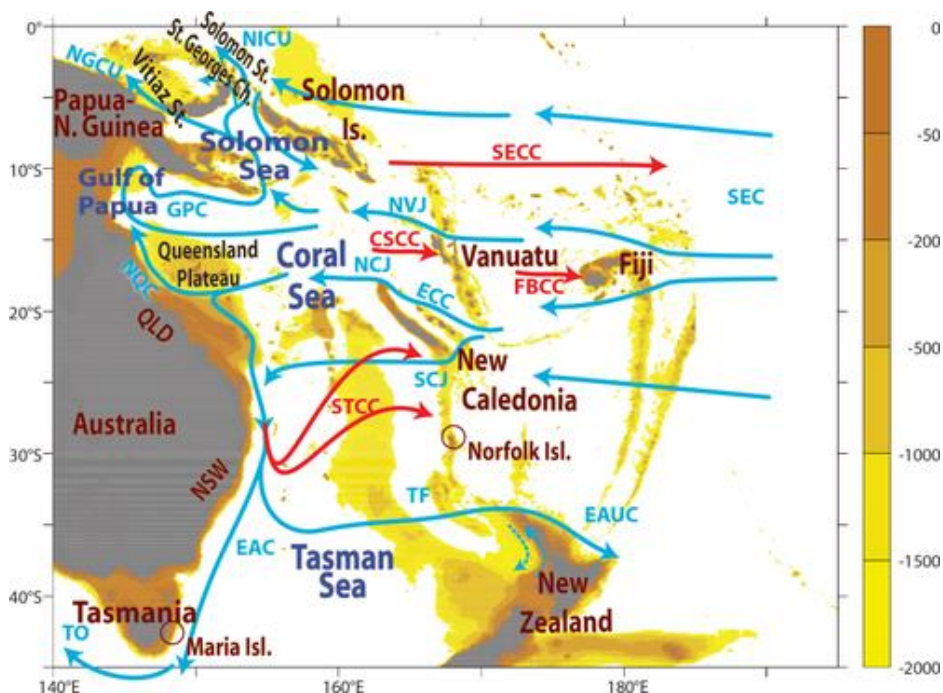


Figure 7-8: The dominant surface ocean currents in the Western Pacific Ocean with the South Equatorial Current (SEC). Where: TF = Tasman Front, EAC = East Australian Current. Depths less than 200 m are shown in yellow-orange colours. Source: Ganachaud et al. (2014).

7.4 Distribution

Shortfin eels are found in Aotearoa-New Zealand, eastern Australia, Tasmania, Norfolk Island, Lord Howe Island, New Caledonia and Rekohu/Chatham islands. Shortfins are abundant in latitudes 35–44°S, although their distribution spans 20–54°S. Shortfin eel can be found at upper altitudes around 250 m southwest of the North Island (around Ruapehu), with occasional records higher (more than 400-500 m; Pike et al. 2019). The upper elevation limit is 1,150 m (Pike et al. 2019).

In Aotearoa-New Zealand, shortfin eels are distributed throughout coastal and low-lying areas of most of the North and South Island (Figure 7-9; Williams et al. 2017). There are few observations of shortfins in Fiordland, but this is likely due to a lack of surveys in this area. In Aotearoa-New Zealand, shortfin eel distribution is best predicted by summer air temperature (as a proxy for summer water temperatures) which explained more than 22% of their distribution (Leathwick et al. 2008).

Shortfin eel larvae have been observed near locations called the Pacific Warm Pool and the low salinity Fresh Pool in the Western Pacific Ocean where they have been captured during research cruises (Kuroki et al. 2020); however, their distribution in the marine environment is poorly understood.

Table 7-1: Summary of key demographic parameters for shortfin eels.

Key demographic parameters	Estimates	References
Maximum size (mm)	1,100–1,200	Jellyman (2013), Williams et al. (2011); Crow and Jellyman (2014)
Minimum size sexual maturity (mm)	400 (Male); 640 (Female)	Fisheries New Zealand (2018)
Age at sexual maturity (yrs)	5–22 (Male); 9–41 (Female)	Fisheries New Zealand (2018)
Maximum longevity (yrs)	60	Jellyman (2013)
Growth parameter (K)	–	–
Fecundity	1.5 to 3 million eggs (500–800 mm length)	Todd (1981)
Larval size _{hatch} (mm)	–	–
Mean egg size (mm)	–	–
Generation length (yrs)	34	Pike et al. (2019)

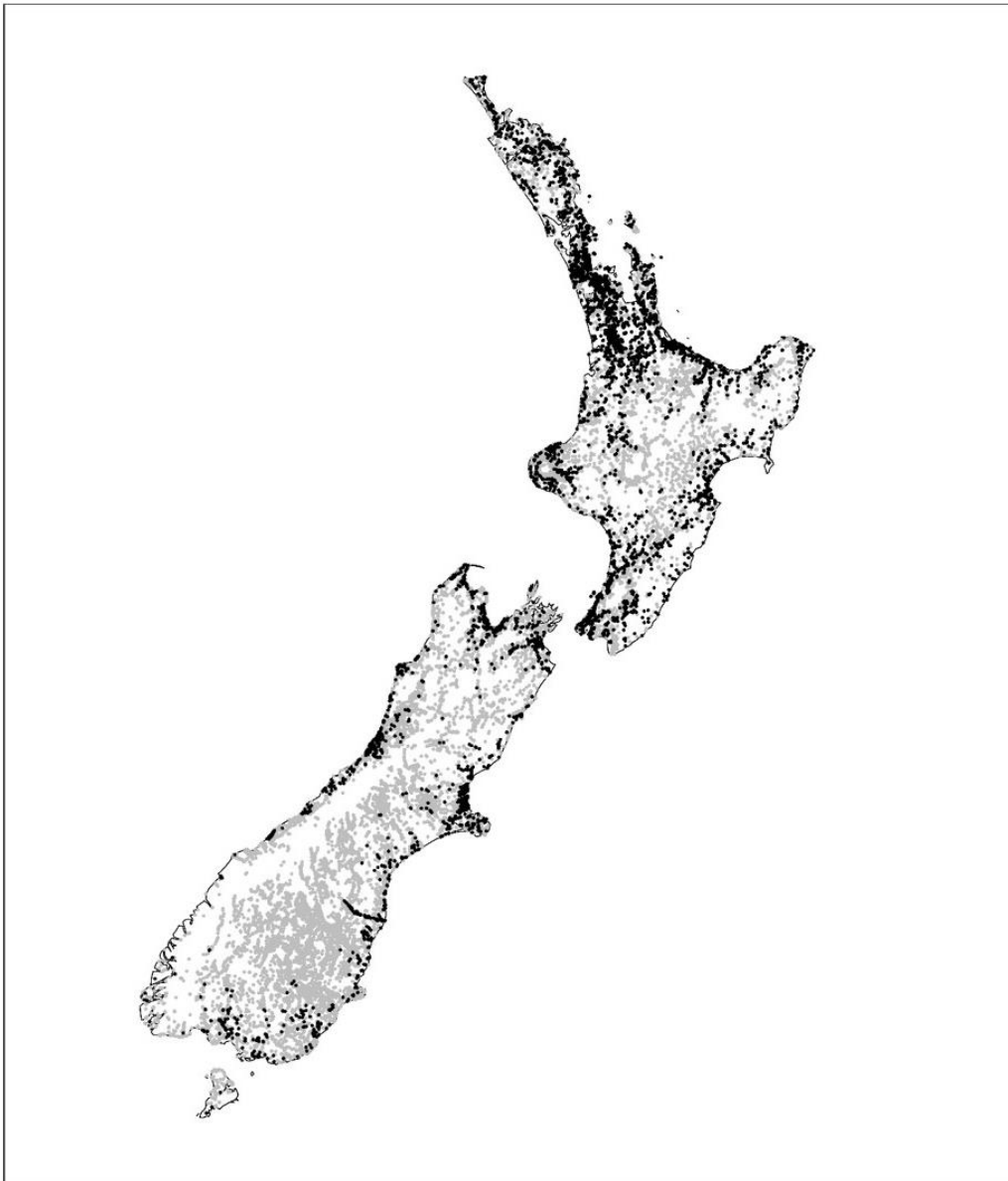


Figure 7-9: Shortfin eel distribution in Aotearoa-New Zealand. Locations of NZFFD records where shortfin eel are present (black circles) and absent (grey circles). Source: NIWA.

7.5 Environmental preferences

The specific environmental preferences of shortfin **larvae** are not well known. The low salinity Fresh Pool is a hydrographic feature in the Western Pacific Ocean that may act as a landmark for shortfin eel spawning areas. The salinity here is typically less than 34 psu, but it is not known if they have a preference for a certain salinity level (Miller et al. 2009). Other environmental preferences such as temperature are not known. The tropical spawning grounds may not be geographically fixed but associated with thermal fronts that can shift within and among years (Miller et al. 2009).

Glass eel recruitment is associated with temperature (August and Hicks 2008). Glass eels migrating upstream in the Tukituki River (Hawkes Bay) showed a clear preference for water temperatures between 12°C and 20°C, with an optimum of 16.5°C. Water temperatures less than 12°C and more

than 22°C almost completely inhibited glass eel migration. In the Waikato River, water temperature and discharge were negatively related to glass eel catches. Both water temperature and discharge had significant inverse relationships with glass eel catches, with temperature explaining more than 30% of the variance in catch periodicity. Other environmental variables associated with glass eel migration include the lunar period. There was a positive correlation between glass eel migrations and spring tides, with peak migration periods typically occurring within a few hours of the peak of high tide, and between 2 and 4 days after the day of spring tide (Jellyman and Lambert 2003). Shortfin glass eels are thought to use olfactory cues to select the river they will enter (McCleave and Jellyman 2002). The chemical composition of the cues shortfin glass eel use is not known but are likely related to the presence of other eels in the specific catchment.

Generally, Aotearoa-New Zealand-wide records indicate that **adult shortfin eel** density is highest in large rivers with stable flows located at low altitude and close to the sea. High summer temperatures are also important. Other factors such as gradient, substrate type and indigenous forest cover tend to be less important in influencing the abundance of shortfin eels. Shortfin eel movements and adult migrations are related to environmental variables especially temperature, rainfall and lunar patterns. Environmental cues for downstream migration of mature adult eels are summarised in Table 7-2. While rainfall is considered an important trigger for downstream migration, the rainfall threshold that triggers movement is unknown (Table 7-2). Temperature does not appear as important as hydrological cues for downstream migration (Table 7-2); however, the environmental triggers/thresholds are poorly known and the correlations are weak. In highly regulated hydro lakes, the relationship between environmental variables and downstream eel migration are poorly described.

Table 7-2: Environmental cues for the downstream migration of Aotearoa-New Zealand eels. Grey shaded areas indicate these variables were not tested/not significant in the study referenced.

Rainfall/flow	Lunar cycle	Pressure changes	Temperature	Reference
Floods	Not moonlit nights			Downes (1918)
Early stages of floods	Not moonlit nights			Best (1929)
Floods				Cairns (1941)
High rainfall	Low catches on full moon	Important		Burnet (1969)
Rainfall	Low catches on full moon	Important with rainfall and dark moon		Todd (1981)
Floods	No effect	No obvious effect	Less than 11°C inhibiting	Boubée et al. (2001)
Rainfall	New moon best			Watene et al. (2003)
Rainfall	No effect			Boubée and Williams (2006)
High lake outflow	No effect			Jellyman and Unwin (2017) Stuart et al. (2018)

Adult shortfin eels have the highest upper thermal limit of native fish species assessed by Olsen et al. (2012). Adult shortfin eels have a critical thermal maximum 39.7°C (at an acclimation temperature of 15°C). The critical thermal maximum for shortfin eel elvers increased from 33.4°C, when acclimated at 12°C, to 38.6°C when acclimated at 30°C. Graynoth and Taylor (2000) reported that growth of shortfin eels ceased below 8.9°C for fish fed on maximum rations. This appears to be the sole estimate of a thermal minimum for growth of an Aotearoa-New Zealand native fish species.

The relationships between shortfin eel growth rates and water temperatures have not been fully resolved even though there are multitude studies correlating growth rates to water/air temperature. Growth appears to be unrelated to mean summer temperatures because there is no latitudinal trend in eel growth rates in Aotearoa-New Zealand (Jellyman 1997). Water temperatures more than 12.4°C have been shown to promote growth rates for shortfin eels in some rivers (Graynoth and Taylor 2004), while feeding shortfins are usually inactive in Te Waihora at water temperature less than 12°C which corresponds to the period between May and September (Jellyman et al. 1996). Once temperatures exceed 12°C, shortfin eel feeding activity increases, especially in spring and early summer. In Te Waihora, shortfin eel growth rates were not associated with air temperature, fish density or food availability (Graynoth and Jellyman 2002) although movement is associated with water temperature (Jellyman et al. 1996). Water temperature affects the length of the growing season (Jellyman 1997) and indirectly affects foraging and feeding activity in eels (Jellyman 1991).

7.6 Diet and predation

The marine diet of shortfin eel larvae is not known. Presumably the larvae feed on marine snow, zooplankton and gelatinous plankton like other eel species who inhabit the marine environment (Riemann et al. 2010; Ayala et al. 2018). Studies have shown the larvae of European eel have a diverse diet that includes cnidaria (jellyfish like organisms), worms (polychaeta) and crustaceans (Figure 7-10).

In fresh water, juvenile and adult shortfin eels are omnivorous and eat a wide diversity of food types, primarily aquatic invertebrates. In lowland rivers, dietary analysis shows that smaller eels (80–100 mm) consume fly-larvae (chironomidae), free-living caddis flies and ostracods (crustaceans). In Te Waihora, juvenile shortfins forage on preyfish (mainly common bullies, *Gobiomorphus cotidianus*) at a smaller size (400 mm) than historically recorded (more than 500 mm; Kelly and Jellyman 2007). Flooded river margins are important feeding grounds for eels. During a flood, eels will move out of the main river channel to the margins where they will fill up organisms not usually found in the water, like earthworms, spiders and beetles (Chisnall 1987; Chisnall 2000).

Comparisons of shortfin eel diets in 1970 (Ryan 1986) with 2007 analyses (Kelly and Jellyman 2007) indicate shortfin eel diets have shifted from benthic biota (i.e., *Potamopyrgus*) to fly larvae (Chironomus species). This shift towards foraging on smaller sediment-dwelling species could have implications for juvenile eel bioenergetics, and may help explain why juvenile shortfin growth rates have significantly decreased in past decades (Kelly and Jellyman 2007). Dietary and stable isotope signatures indicated that small shortfins (100–299 mm) have considerable overlap in trophic position with common bullies, the dominant fish in Te Waihora (92% of total abundance). This indicates that these two species may directly compete for food resources and highlights the importance of Chironomus species in sustaining eel populations in Te Waihora (Kelly and Jellyman 2007).

Ontogenetic shifts in diet have been observed in shortfin eels. Medium size eels (100–199 mm total length) eat fly larvae and several species of uncased caddis flies, while larger eels (200–300 mm total length) consume cased caddis flies and ostracods (Sagar and Glova 1998). In lakes, shortfin eels are opportunistic scavengers and will forage along inundated shorelines during high water levels and will

consume terrestrial organisms (Jellyman 1989). Feeding eels also move with the wind, possibly taking advantage of wind-derived currents (Jellyman and Smith 2008). Snails (*Potamopyrgus*) and fish species such as common bullies are an important component of eel diets in lake populations (Jellyman 1989; Rowe 1999), but kākahi are important in other lakes such as Horowhenua (Chisnall and Jellyman 1999). Hayes and Rutledge (1991) demonstrated a switch in the diets of several fish species, including shortfin eels, from being predominantly benthic to pelagic (specifically mysids) between clear and turbid arms of Lake Waahi and Whangape.

Predators of shortfin eel larvae in the marine environment are unknown. Mesopelagic fishes (living between 0–300 m deep) especially lanternfish (Myctophidae) predate on the larvae of the European eel (Jensen et al. 2018) and may also predate on shortfin eel larvae (Flynn and Marshall 2013). Predators of glass eels in coastal and estuarine waters are not known. Elvers are eaten by several bird species including the Australasian bittern¹¹, Hochsetters frogs (Najera-Hillman et al. 2009) and by larger longfin and shortfin eels (e.g., Sagar and Glova 1998).

¹¹ <http://nzbirdsonline.org.nz/species/australasian-bittern>

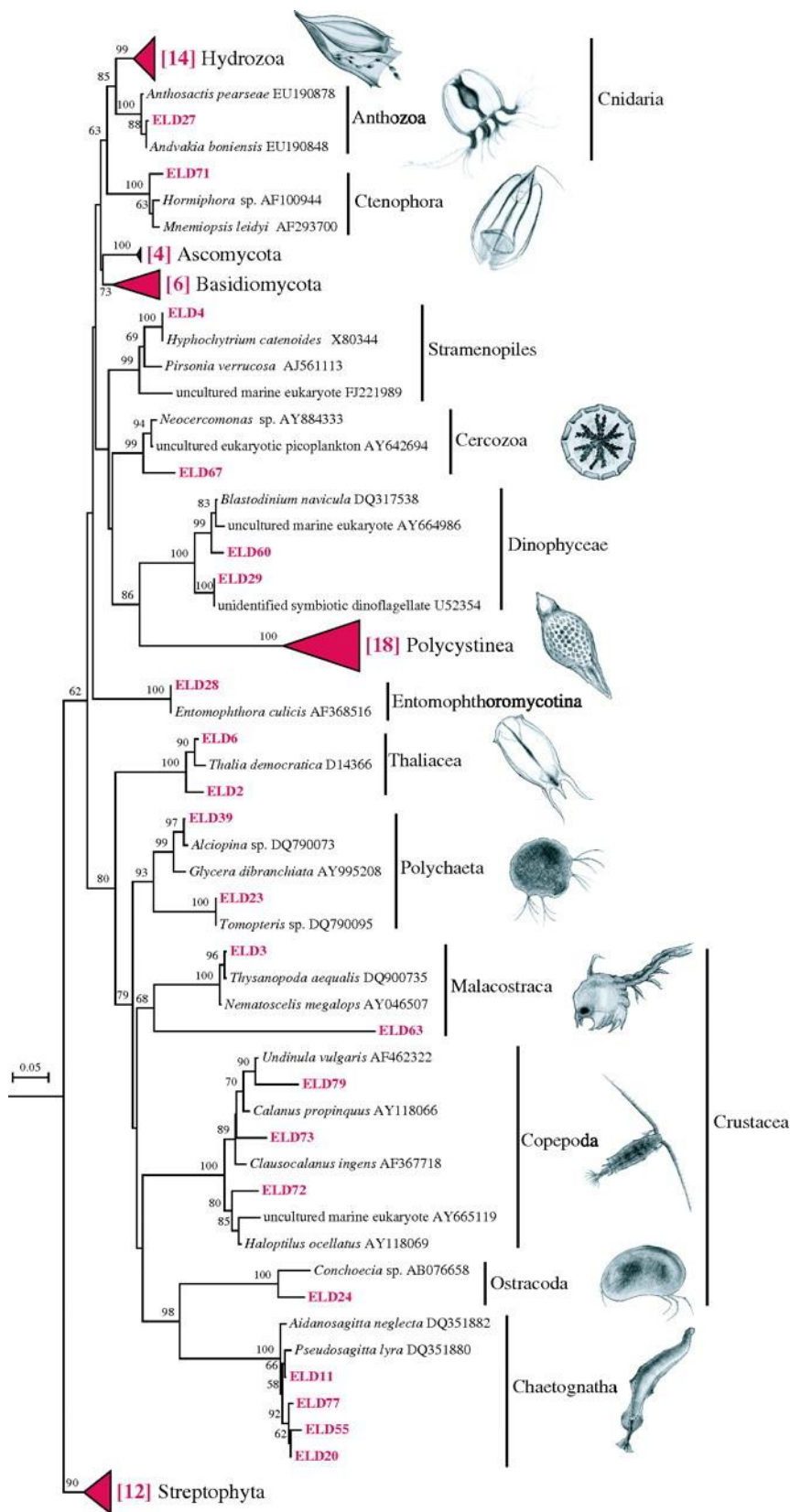


Figure 7-10: Diet of European eel larvae during marine life. Source: Riemann et al. (2010).

7.7 Recruitment

Compared to the Northern Hemisphere, there are relatively few glass eel or elver long-term recruitment datasets available in Aotearoa-New Zealand. Studies investigating recruitment trends for Aotearoa-New Zealand eels includes a variety of research approaches conducted at a range of locations, over various time periods– including glass eel recruitment patterns, numbers of elvers arriving at hydro stations, age class structure of juvenile eels, and length frequency data from commercial catch sampling. Using the age class structure of juvenile eels there is evidence that glass eel recruitment has declined in two North Island and three South Island catchments. While other studies on the variability and temporal abundance of glass eels over a seven-year period from 1995 to 2002 showed no decline in recruitment for either species¹².

The recruitment of glass eels shows considerable year-to-year variation. For example, in the 1970's, numbers of glass eels caught in consecutive years varied by a factor of 10 (Jellyman 1979). Jellyman et al. (1999) recorded that October was the main month for arrival of glass eels of both species on the west coast of the South Island, although catches were higher during September in one of the study years. On the Tukituki River (Hawkes Bay), almost three times more longfin and shortfin glass eels (50,287) were caught in 2001 compared to 2002 (19,954) (August and Hicks 2008) indicating significant variation in recruitment can occur among years.

In the Waikato River, glass eel recruitment (predominantly shortfin eels) was studied over a period of two years (2004–05; Jellyman et al. 2009). Comparison of catch results from this study with previous catch data (30 years apart), found glass eel migration periods occurred earlier, by several weeks, in 2004–05 (Jellyman et al. 2009). Jellyman et al. (2009) suggested that the reduced CPUE and duration of the 2004–05 glass eel run indicated that a reduction in recruitment has occurred over time. One explanation given by Jellyman et al. (2009) was that these changes were possibly a result of climate change. Glass eels sampled by Jellyman et al. (2009) were caught between temperatures of 12.6°C and 13.1°C, substantially below the 16.5°C optimum suggested by August and Hicks (2008). There was also evidence that glass eels may compensate for increased temperature through earlier seasonal migration (Jellyman et al. 2009). Therefore, if they were to wait for river temperatures more than 12°C before beginning upstream migration, the time window would become limited (Jellyman et al. 2009).

Length-frequency distributions have also been used to investigate changes in recruitment. For example, Jellyman and Chisnall (1999) found a bimodal size distribution in Te Waihora with the 200–300 mm size classes being poorly represented. This probably reflects poor recruitment over several years, either because lake opening times did not coincide with the availability of glass eels, or because overall numbers of glass eels coming in when the lake was open were low.

Shortfin elver recruitment is monitored at various elver trap-and-transfer sites around Aotearoa-New Zealand. These studies have shown that elver abundances can vary two-fold between some years, but that overall there has been no trends in recruitment (Figure 7-11). Elvers arriving at Matahina Dam (Rangitāiki River) and Karāpiro Dam (Waikato River) have both show similar annual patterns over the 15 years of continuous data collection. While shortfin recruitment at the Pātea Dam (Pātea River) was observed to decrease between 2005 and 2011, but has steadily increased each year after this. In contrast, Piripaua (Waikaretaheke River) shortfin catches have remained very stable during the early 2000's and have increased markedly over the last five years. Data across all sites were used by Martin

¹² https://fs.fish.govt.nz/Doc/21722/25_EEL_09.pdf.ashx

and Bowman (2016) to conclude that there has been no decline in elver recruitment at selected hydro dams over the last 25 years.

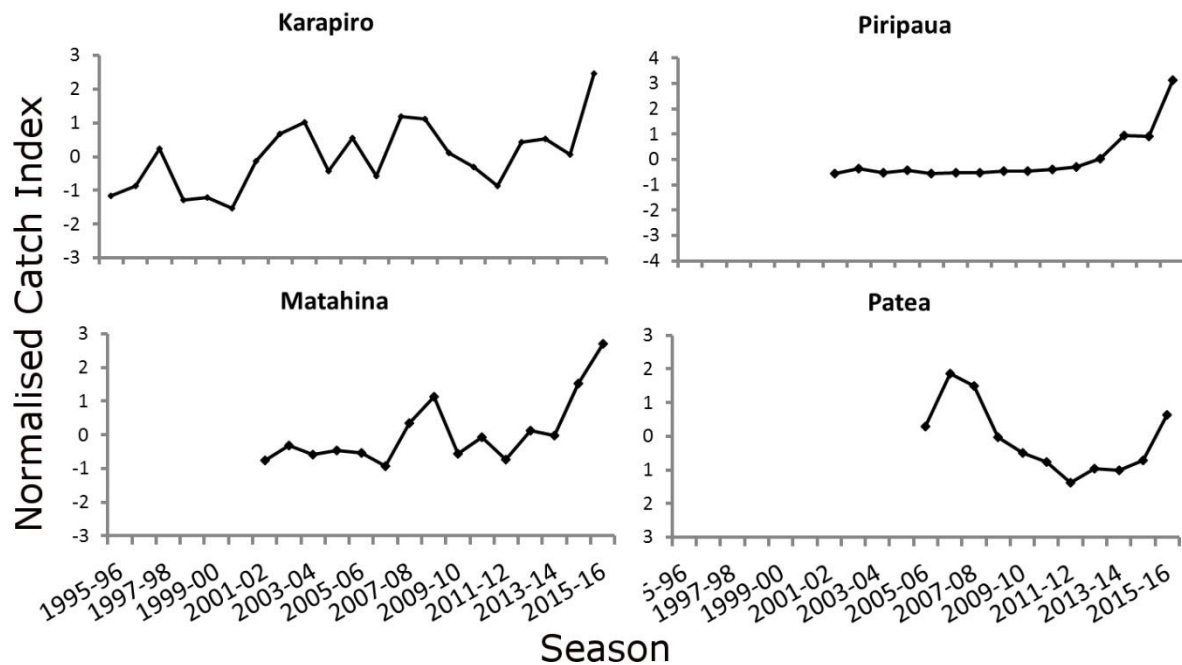


Figure 7-11. Normalised (see Durif et al. 2008) index of relative abundance for shortfin elver catches from 1995–96 to 2015–16. Source: Martin and Bowman (2016).

7.8 Current impacts of climate change

Observations indicate that extreme climatic events, namely El Niño and La Niña, droughts and storms can affect shortfin eels. Droughts are recognised as an ongoing threat to shortfin eels according to the IUCN assessment (Pike et al. 2019).

7.8.1 El Niño/La Niña (and Southern Oscillation Index)

During the extreme 1997 El Niño event, commercial eel harvests in the North Island were severely reduced (Chisnall and Kemp 1998). High water temperatures and low rainfall meant there was a lack of freshes/floods to stimulate eel movement and therefore the probability of capture declined. In Te Waihora, shortfin eel growth rates were related to environmental variables over a 5-year period (1995–1999; Graynoth and Jellyman 2002). Although not statistically significant, the highest growth rates were observed in the warmest summer (1995–1996) and the lowest in the coldest summer (1996–1997) suggesting temperature affects shortfin eel growth rates. However, the lack of long-term data spanning multiple years and decades made it difficult to track changes in growth rates with respect to environmental variation and a changing climate. Nevertheless, the period of analysis coincides with the significant El Niño event in 1997.

In three streams (Pigeon Bay [Banks Peninsula]; Horokiwi [Wellington] and Te Maari [Waikato]), shortfin growth rates declined by 1.4 mm per year over the period 1996 to 1998 again coinciding with an El Niño event (Graynoth and Taylor 2004). These declines were attributed to annual variation in weather and streamflow between 1996 and 1998 but were not explicitly incorporated into any statistical analysis (Graynoth and Taylor 2004).

Chisnall et al. (2002) found variations in the size and recruitment patterns of shortfin and longfin glass eel that appeared to be related to El Niño events in 1997 and 1998. There is evidence to suggest that the El Niño Southern Oscillation (ENSO) and associated changes to ocean conditions may influence eel recruitment in the western South Pacific (Jellyman and Bowen 2009). Between 1995 and 1998, glass eel recruitment was monitored in 13 streams on the east and west coasts of Aotearoa-New Zealand (Jellyman et al. 2002) again coinciding with the 1997 El Niño. Annual variation in recruitment was found and was hypothesised to be influenced by the El Niño Southern Oscillation, with greater recruitment occurring in the north-west during the El Niño phase (when there are persistent westerly winds). During La Niña when the wind direction is more persistent from the north-east, greater recruitment was observed in the north-east. The Antarctic Circumpolar Wave¹³ which effects how ENSO cycles develop could also provide a further mechanism for the periodic alteration of glass eel recruitment in Aotearoa-New Zealand¹⁴. The mean length of glass eels decreased in all regions between 1997 and 1998 which was hypothesised to a result of changes in the duration of the migration from their spawning grounds (Jellyman et al. 2002). In the Waikato River, glass eel recruitment (predominantly shortfin eels) was examined in 2004 and 2005 (Jellyman et al. 2009). Comparison of catch results from this study with previous catch data (30 years apart), found glass eel migration periods are occurring several weeks earlier and it was hypothesized that this may be due to climate change (Jellyman et al. 2009).

7.8.2 Drought

The latest IUCN threat assessment identifies drought as a significant ongoing threat to shortfin eels that affects 50–90% of the population (Pike et al. 2019). In Aotearoa-New Zealand, significant eel mortalities have been observed in recent years in several locations including Lake Tūtira¹⁵ (Napier; Figure 7-12) and the Selwyn River¹⁶ (Canterbury). In Lake Tūtira, the pH was greater than 9, while in the Selwyn River, prolonged drought conditions and lack of winter rainfall for three consecutive years were hypothesised to be associated with eel mortalities¹⁷.



Figure 7-12: Dead eels along the shoreline of Lake Tūtira, Napier. Source: Stevie Smith, Hawke’s Bay Regional Council.

¹³ The Antarctic Circumpolar Wave is a coupled [ocean/atmosphere wave](#) that circles the [Southern Ocean](#) in approximately eight years at 6–8 cm/s.

¹⁴ https://fs.fish.govt.nz/Doc/21722/25_EEL_09.pdf.ashx

¹⁵ <https://www.stuff.co.nz/environment/87628341/eel-not-be-right-mass-fish-deaths-at-lake-under-investigation>

¹⁶ <https://www.stuff.co.nz/environment/86992441/road-or-river-barren-selwyn-reaches-new-low-swimming-spot-stagnant>

In the upper Waipaoa catchment (Gisborne), it was hypothesised that eel recruitment was limited by drought conditions (Ruru and Chisnall 2007). In the Ashley River (North Canterbury) eels and many other fish species are salvaged and relocated because of dry conditions. Reduced shortfin eel catches have been observed in the North Island during exceptionally dry (Beentjes 2019).

In Australia, during the “Millennium Drought” (1990s–2000s), commercial shortfin eel catches declined because harvestable waters reduced (McKinnon 2002; VFA 2017). Following several years of prolonged drought, significant eel mortalities were found in several lakes in Australia in 2006 and 2007 (Leahy et al. 2007). High evaporation rates resulted in a rapid increase in lake salinity which altered the salinity levels in eel blood plasma and ultimately reached lethal concentrations. The lethal effect of increased salinity on eels was magnified when the pH was greater than 9.

7.8.3 Storms

In 1968, the Wahine storm destroyed the macrophyte beds along the shoreline of Te Waihora/Lake Ellesmere which resulted in an ecosystem regime shift (Figure 7-13). Te Waihora effectively transitioned from a clear macrophyte-dominated state to a turbid phytoplankton dominated state which subsequently altered the structure of the food web. Although the Wahine storm is not explicitly related to climate change, more frequent storm events are anticipated with climate change (MFE 2008) which may change the ecosystems of shallow coastal lakes such as Te Waihora.

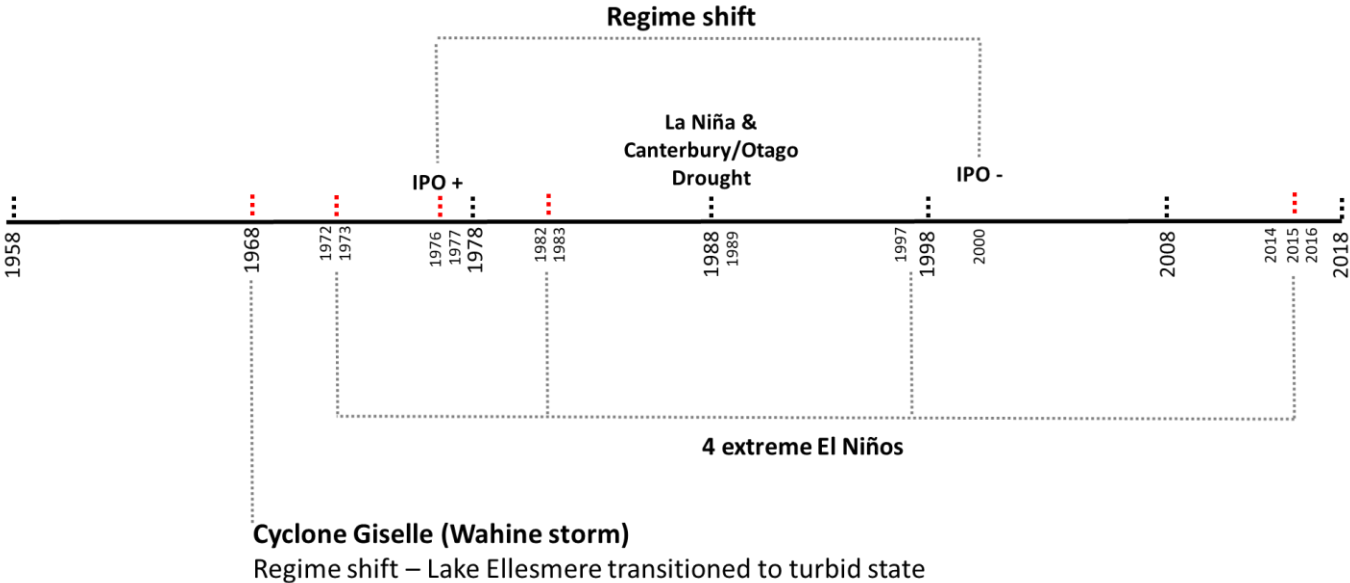


Figure 7-13: Timeline of significant climate events in Te Waihora from 1958 to present. Source: Egan (unpublished data).

An on-going piece of research is reconstructing the growth rates of shortfin eels from Te Waihora using otoliths collected from 1979 by Todd (1981) to 2014. This is one of the first studies in Aotearoa-New Zealand to explicitly investigate the relationship between eel growth rates and climate/environmental variables over multiple decades. The growth patterns show years with higher (i.e., 1973) and lower growth rates (i.e., 1970; Figure 7-14). The initial results indicate that increased wind speeds that may be associated with storms reduced shortfin eel growth rates. These results are tentative, and more analyses are being done.

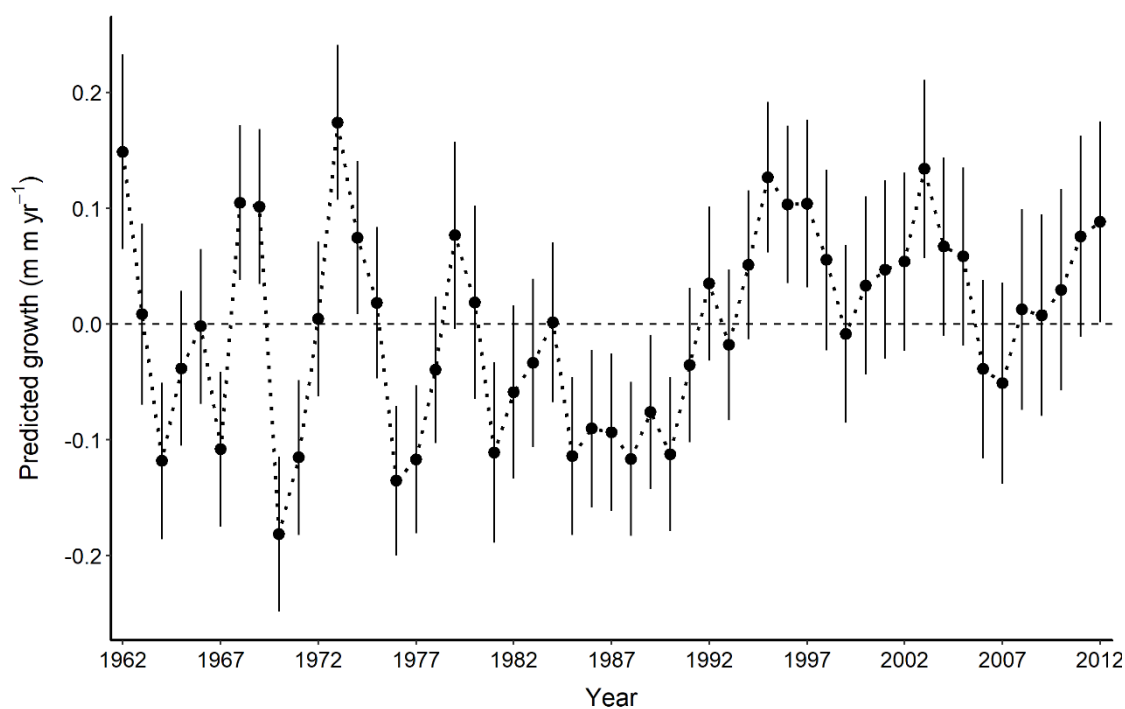


Figure 7-14: Time series of shortfin eel growth rates in Te Waihora/Lake Ellesmere from 1962–2012. The horizontal dashed line is the model intercept and the vertical lines are the 95% confidence intervals of the predicted values. Source: Egan (unpublished data).

7.8.4 Temperature

Eel catches are greatly influenced by water temperature, flood events (increased catches) and drought conditions (reduced catches). Catches decline in winter months (May to September), particularly in the South Island, where fishing ceases. Although water temperature did not influence catch rates (CPUE, catch-per-unit-effort) of either size group, catches of the smaller eels were greater during the new moon phase than during the other phases—catches of larger eels were unaffected by lunar phase (Jellyman and Chisnall 1999). Other information to consider is that Graynoth and Taylor (2012) found that food availability as opposed to temperature was the most significant driver of annual growth rate variation in shortfin eels. However, little is known about variation in eel food resources, freshwater food webs and how this may be related to climate change.

In Australia, the distributions of shortfin eels in rivers and streams are predicted to change with climate change (Bond et al. 2011). The probability of shortfin eel occurrence in rivers across Victoria was linked with physiographic, climatic, and hydrologic characteristics. This was done for baseline (historic) and future climate scenarios which corresponded to temperature increases (by 2030) of +0.54°C, +0.85°C and +1.24°C, respectively. Results indicated that at +1.2°C by 2030, there would be a 25% reduction in shortfin eel occurrence while at +0.85°C shortfin eel occurrence would be reduced by 20% (Bond et al. 2011).

It is postulated that declining glass eel recruitment is a result of climate change. Although no consistent national trend exists, freshwater temperatures in Aotearoa-New Zealand have increased by ~1.6°C from 1989–2005 (Scarsbrook 2006 in Jellyman et al. 2009).

An analysis of shortfin elver catches at Karāpiro Dam has shown that the numbers of elvers caught each day increases as the water gets warmer though summer. Similarly, daily catches of shortfin elvers

decline as winter approaches and the water cools. You can see this in Figure 7-15 where the daily catches of elvers increase throughout December as the water temperature increases (see the solid blue line in Figure 7-15 which is the average daily catch over 20 years) and then declines in March-April as summer finishes and the water cools.

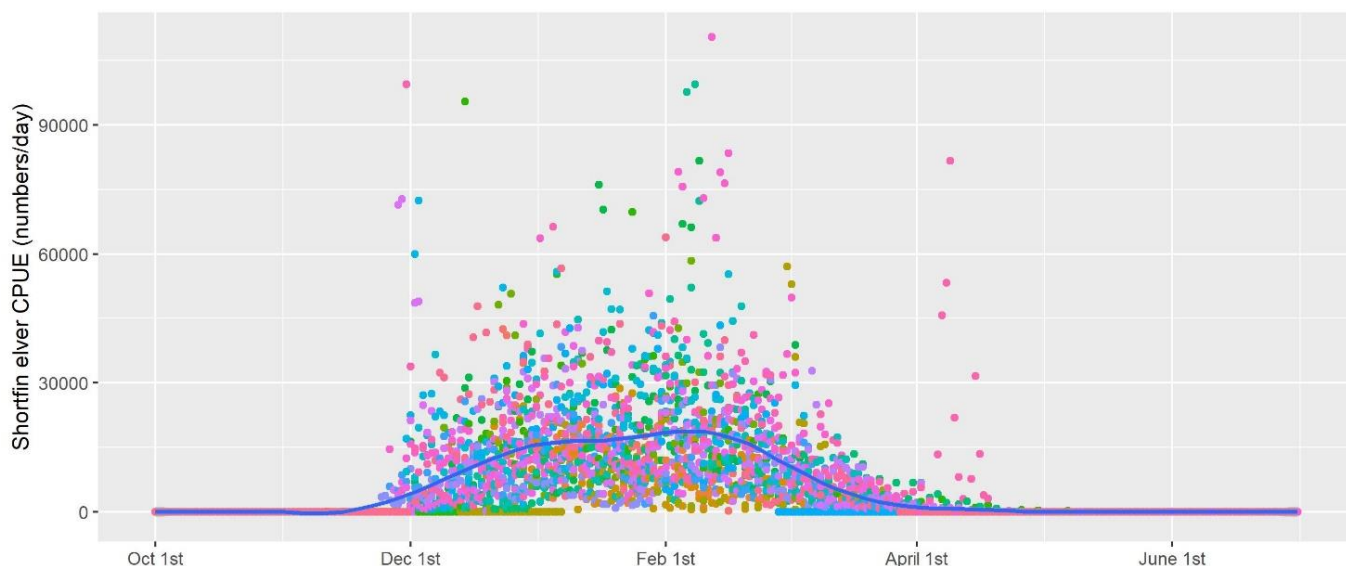


Figure 7-15: The number of shortfin elvers caught each day (known as catch-per-unit-effort or CPUE) at Karāpiro Dam over 20 years. Each dot indicates the number of elvers captured on each day while the different colours indicate different years. The solid blue line shows the average daily catches of shortfin elvers over the 20 years. Source: Shannan Crow, unpublished.

7.9 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between shortfin eel and the wider ecosystem include:

- As a predator and prey species, where smaller eels are predated upon by larger eels and birds (e.g., cormorants, herons). The glass eel stage is particularly rich in lipids and may be an important food source for coastal/estuarine species;
- Eels play an important part in recycling nutrients in freshwater ecosystem (Dolamore et al. 2017). Shortfin eels contribute marine subsidies to coastal, estuarine and freshwater ecosystems;
- The toxic algae *Nodularia spumigena* can be found in the liver and muscle of shortfin eels from Lake Forsyth/Te Wairewa and can pose risks to human consumption (Dolamore et al. 2017).

7.10 Additional multiple stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). The stressors derived from the IUCN threat assessment (n=4), included ‘fishing and harvesting of aquatic resources [large scale harvest]’; ‘dams and water management/use [abstraction of surface water and dams]’; and ‘climate change and severe weather [droughts]’ (see Supplementary Appendix 2).

7.11 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- As described in Pike et al. (2019), under the New Zealand Threat Classification Series, shortfin eel has been assessed as ‘Not Threatened’ (species with large stable populations), with qualifier ‘Increasing’ (Dunn et al. 2018). The New Zealand Threat Classification Series defines population as “The total number of individuals that are resident or that breed in New Zealand” referring solely to mature individuals (Townsend et al. 2008). For a ‘Not Threatened’ taxon, there must be 20,000–100,000 or >100,000 mature individuals, and the population trend must be stable ($\pm 10\%$) or increasing ($>10\%$; Townsend et al. 2008). The qualifier ‘Inc’ refers to an ongoing or predicted increase of $>10\%$ in the total population taken over the next 10 years or 3 generations, whichever is longer (Townsend et al. 2008). The difficulty with assessing an anguillid species using this terminology, as with the IUCN classification, is that there is no direct metric for ‘mature individuals’, as eels are not considered mature until they reach breeding grounds to spawn. Therefore, it is necessary to assess them using a suitable alternative metric (for example, silver eel escapement/glass eel recruitment);
- As summarised by Allibone et al. (2009), for species for which key aspects of their life histories are unknown (such as longfin eels), it is difficult to understand what is driving any population declines. A critical knowledge gap is that it is unknown if a minimum number of eels are required to reach the spawning grounds to achieve successful spawning and recruitment (Allibone et al. 2009). There exists a risk that a minimum number of longfin eels are required to migrate in any year to ensure eels encounter one another at the ocean spawning sites.

7.12 References

- Allibone, R., David, B., Hitchmough, R., Jellyman, D., Ling, N., Ravenscroft, P., Waters, J. (2010) Conservation status of New Zealand freshwater fish, 2009. *New Zealand Journal of Marine and Freshwater Research*, 44: 271–287.
- Arai, T., Otake, T., Jellyman, D.J., Tsukamoto, K. (1999) Differences in the early life history of the Australasian shortfinned eel *Anguilla australis* from Australia and New Zealand, as revealed by otolith microstructure and microchemistry. *Marine Biology*, 135: 381–389.
- August, S.M., Hicks, B.J. (2008) Water temperature and upstream migration of glass eels in New Zealand: implications of climate change. *Environmental Biology of Fishes*, 81(2): 195–205.
- Ayala, D.J., Munk, P., Lundgreen, R.B.C., Traving, S.J., Jaspers, C., Jørgensen, T.S., Hansen, K.H., Riemann, L. (2018) Gelatinous plankton is important in the diet of European eel (*Anguilla anguilla*) larvae in the Sargasso Sea. *Scientific Reports*, 8: 6156.
- Beentjes, M.P. (2019) Monitoring commercial eel fisheries: 2015–16 to 2017–18. *New Zealand Fisheries Assessment Report 2019/42*. Fisheries New Zealand, Wellington: 92. <https://fs.fish.govt.nz/Doc/24740/FAR-2019-42-monitoring-commercial-eel-fisheries.pdf.ashx>

- Best, E. (1929) Fishing methods and devices of the Maori. *Dominion Museum Bulletin*, 12: 1-230.
- Bond, N.R., Thomson, J.R., Reich, P. (2014) Incorporating climate change in conservation planning for freshwater fishes. *Diversity and Distributions*, 20: 931–942.
- Boubée, J.A.T, Williams, E.K. (2006) Downstream passage of silver eels at a small hydroelectric facility. *Fisheries Management and Ecology*, 13: 165-176.
- Boubée, J.A.T, Mitchell, C.P., Chisnall, B.L., Bowman, E.J., Haro, A. (2001) Factors regulating the downstream migration of mature eels (*Anguilla* spp.) at Aniwhenua Dam, Bay of Plenty, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 35: 121–134.
- Burnet, A.M.R. (1969) Migrating eels in a Canterbury river, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 3: 230–243.
- Cairns, D. (1941) Life-history of the two species of New Zealand freshwater eel. Part I. Taxonomy, age, growth, migration, and distribution. *New Zealand Journal of Science & Technology*, 23: 53–72.
- Chisnall, B.L. (1987) *Juvenile Eel Biology in the Backwaters of the Waikato River*. MSc thesis, University of Waikato, New Zealand.
- Chisnall, B.L. (2000) Utilisation of flood plains by New Zealand freshwater eels. *NIWA Client Report: CEE01201*. Report prepared for Central Eel Enhancement Co Ltd: 15.
- Chisnall, B.L., Kemp, C. (1998) Size, age, and species composition of commercial eel catches from market sampling in the North Island (1997-98). Ministry of Fisheries Research Project EEL9701 Objective 2: 73. <https://fs.fish.govt.nz/Doc/22481/EEL9701%20Eel%20North%20Island%20size%20age%20species%20final%20Objective%202.pdf.ashx>
- Chisnall, B.L., Jellyman, D.J. (1999) Synoptic fisheries surveys of Lake Horowhenua. *NIWA Client Report: TOW90501* 34 p. Report prepared for Te Ohu Kaimoana.
- Chisnall, B.L., Jellyman, D.J., Bonnett, M.L., Sykes, J.R. (2002) Spatial and temporal variability in length of glass eels (*Anguilla* spp.) in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 36: 89–104. DOI: 10.1080/00288330.2002.9517073.
- Clements & Associates Limited (2016) *The Atlas of Area Codes & TACCs 2016-17*. Clements & Associates Limited Report.
- Crow, S.K., Jellyman, D.J. (2014) Assessment of the eel fishery in the Pouto Lakes in Northern Kaipara. *New Zealand Fisheries Assessment Report 2014/21*. April 2014: 40.
- Davey, A J.H., Jellyman, D.J. (2005) Sex Determination in Freshwater Eels and Management Options for Manipulation of Sex. *Reviews in Fish Biology and Fisheries*, 15: 37–52. DOI: 10.1007/s11160-005-7431-x.
- Dolamore, B., Puddick, J., Wood, S.A. (2017) Accumulation of nodularin in New Zealand shortfin eel (*Anguilla australis*): potential consequences for human consumption. *New*

Zealand Journal of Marine and Freshwater Research, 51: 321–332, DOI: 10.1080/00288330.2016.1236738

- Downes, T.W. (1918) Notes on eel and eel-weirs (tuna and pa-tuna). *Transactions and Proceedings of the New Zealand Institute*, 50: 296–316.
- Dunn, N.R., Allibone, R.M., Closs, G.P., Crow, S.K., David, B.O., Goodman, J.M., Griffiths, M., Jack, D.C., Ling, N., Waters, J.M., Rolfe, J.R. (2018) Conservation status of New Zealand freshwater fishes, 2017. Department of Conservation: 11.
- Fisheries New Zealand (2018) Fisheries Assessment Plenary. Stock Assessment and Stock Status Volume 1: Introductory Section to Groper. Fisheries New Zealand, Wellington. May 2018: 511.
- Flynn, A.J., Marshall, N.J. (2013) Lanternfish (Myctophidae) Zoogeography off Eastern Australia: A Comparison with Physicochemical Biogeography. *PLoS ONE*, 8: e80950.
- Foden, W.B., Young, B.E., Akçakaya, H.R., Garcia, R.A., Hoffmann, A.A., Stein, B.A., Thomas, C.D., Wheatley, C.J., Bickford, D., Carr, J.A., Hole, D.G., Martin, T.G., Pacifici, M., Pearce-Higgins, J.W., Platts, P.J., Visconti, P., Watson, J.E.M., Huntley, B. (2019) Climate change vulnerability assessment of species. *WIREs Climate Change*, 10: e551.
- Ganachaud, A., Cravatte, S., Melet, A., Schiller, A., Holbrook, N.J., Sloyan, B.M., Widlansky, M J., Bowen, M., Verron, J., Wiles, P., Ridgway, K., Sutton, P., Sprintall, J., Steinberg, C., Brassington, G., Cai, W., Davis, R., Gasparin, F., Gourdeau, L., Hasegawa, T., Kessler, W., Maes, C., Takahashi, C., Richards, K.J., Send, U. (2014) The Southwest Pacific Ocean circulation and climate experiment (SPICE). *Journal of Geophysical Research-Oceans*, 119: 7660–7686.
- Graynoth, E., Jellyman, D.J. (2002) Growth, survival, and recruitment of juvenile shortfinned eels (*Anguilla australis*) in a large New Zealand coastal lake, *New Zealand Journal of Marine and Freshwater Research*, 36: 25–37. DOI: 10.1080/00288330.2002.9517068.
- Graynoth, E., Taylor M.J. (2004) Growth of juvenile eels (*Anguilla* spp.) in lowland streams in New Zealand. *Fisheries Research*, 66: 95–106.
- Hayes, J.W., Rutledge, M.J. (1991) Relationships between turbidity and fish diets in Lakes Waahi and Whangape, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 25: 297–304.
- Jensen, M.R., Knudsen, S.W., Munk, P., Thomsen, P.F., Moller, P.R. (2018) Tracing European eel in the diet of mesopelagic fishes from the Sargasso Sea using DNA from fish stomachs. *Marine Biology*, 165: 130. <https://doi.org/10.1007/s00227-018-3390-3>
- Jellyman, D.J. (1979) Upstream migration of glass eels (*Anguilla* spp.) in the Waikato River. *New Zealand Journal of Marine and Freshwater Research*, 13: 13–22.
- Jellyman, D.J. (1989) Diet of two species of freshwater eel (*Anguilla* spp.) in Lake Pounui, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 23: 1-10.
- Jellyman, D.J. (1991) Factors affecting the activity of two species of eel (*Anguilla* spp.) in a small New Zealand lake. *Journal of Fish Biology*, 39: 7–14.

- Jellyman, D.J., Glova, G.J., Todd, P.R. (1996) Movements of shortfinned eels, *Anguilla australis*, in Lake Ellesmere, New Zealand: Results from mark-recapture studies and sonic tracking. *New Zealand Journal of Marine and Freshwater Research*, 30: 371–381. DOI: 10.1080/00288330.1996.9516724
- Jellyman, D.J. (1997) Variability in growth rates of freshwater eels (*Anguilla* spp.) in New Zealand. *Ecology of Freshwater Fish*, 6(2): 108–115.
- Jellyman, D.J., Chisnall, B.L., Bonnet, M.L., Sykes, J.R.E. (1999) Seasonal arrival patterns of juvenile freshwater eels (*Anguilla* spp.) in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 33: 249–261.
- Jellyman, D.J., Chisnall, B.L. (1999) Habitat preferences of shortfinned eels (*Anguilla australis*), in two New Zealand lowland lakes. *New Zealand Journal of Marine and Freshwater Research*, 33: 233–248.
- Jellyman, D.J., Chisnall, B.L., Sykes, J.R.E., Bonnett, M.L. (2002) Variability in spatial and temporal abundance of glass eels (*Anguilla* spp.) in New Zealand waterways. *New Zealand Journal of Marine and Freshwater Research*, 36: 511–517.
- Jellyman, D.J., Sykes, J.R. (2003) Diel and seasonal movements of radio-tagged freshwater eels, *Anguilla* spp., in two New Zealand streams. *Environmental Biology of Fishes*, 66(2): 143–154.
- Jellyman, D.J., Smith, C. (2008) Native Fish and Fisheries. In: Hughey KFD, Taylor KJW. (Eds). *Te Waihora/ Lake Ellesmere: The 2007 state of the lake and management futures report*. EOS Ecology, Christchurch.
- Jellyman, D.J., Bowen, M.M. (2009) Modelling Larval Migration Routes and Spawning Areas of Anguillid Eels of New Zealand and Australia. *American Fisheries Society Symposium*, 69: 255–274.
- Jellyman, D.J., Booker, D.J., Watene, E. (2009) Recruitment of *Anguilla* spp. glass eels in the Waikato River, New Zealand. Evidence of declining migrations? *Journal of Fish Biology*, 74: 2014–2033.
- Jellyman, D.J. (2013) The status of longfin eels in New Zealand - an overview of stocks and harvest. *NIWA Client Report: CHC2012-006*. January 2013 (revised): 76.
- Jellyman, D.J., Unwin, M.J. (2019) Fine-scale swimming movement and behaviour of female silver eels, *Anguilla dieffenbachii*, within a lake affected by hydro-power generation. *Fisheries Management and Ecology*, 26: 57–69.
- Jensen, M.R., Knudsen, S.W., Munk, P., Thomsen, P.F., Moeller, P.R. (2018) Tracing European eel in the diet of mesopelagic fishes from the Sargasso Sea using DNA from fish stomachs. *Marine Biology*, 165: 130. DOI: <https://doi.org/10.1007/s00227-018-3390-3>
- Kelly, D.J., Jellyman, D.J. (2007) Changes in trophic linkages to shortfin eels (*Anguilla australis*) since the collapse of submerged macrophytes in Lake Ellesmere, New Zealand. *Hydrobiologia*, 579: 161–173. <https://doi.org/10.1007/s10750-006-0400-0>

- Kuroki, M., Miller, M.J., Feunteun, M., Sasal, P., Pikering, T., Han, Y.-S., Faliex, E., Acou, A., Dessier, A., Schabetsberger, R., Watanabe, S., Kawakami, T., Onda, H., Higuchi, T., Takeuchi, A., Shimizu, M., Hewavitharane, C.A., Hagihara, S., Taka, T., Kimura, S., Mochioka, N., Otake, T., Tsukamoto, K. (2020) Distribution of anguillid leptocephali and possible spawning areas in the South Pacific Ocean. *Progress in Oceanography*, 180: 102234. <https://doi.org/10.1016/j.pocean.2019.102234>
- Leahy, P., Pattern, R., Leonard, A. (2007) Scientific investigation into eel deaths in Western Victoria. *Publication 1173*. EPA, Victoria, Australia: 36.
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- Martin, M., Bowman, E. (2016) Recruitment of freshwater eels 1995–2015. *New Zealand Fisheries Assessment Report, 2016/46*. August 2016: 106. Available via: <https://www.mpi.govt.nz/document-vault/13873>
- McCleave, J., Jellyman, D. (2002) Discrimination of New Zealand stream waters by glass eels of *Anguilla australis* and *Anguilla dieffenbachii*. *Journal of Fish Biology*, 61: 785–800.
- McKinnon, L. (2002) *Victorian Eel Fishery Management Plan*. Department of Natural Resources and Environment, Melbourne.
- Miller, M.J., Kimura, S., Friedland, K.D., Knights, B., Kim, H., Jellyman, D.J., Tsukamoto, K. (2009) Review of ocean-atmospheric factors in the Atlantic and Pacific oceans influencing spawning and recruitment of anguillid eels. Pages 231-249 *In Challenges for Diadromous Fishes in a Dynamic Global Environment. American Fisheries Society Symposium*.
- Ministry for Fisheries (MFish) (2009) *Fisheries Assessment Plenary. Stock Assessments and Stock Status*. Ministry of Fisheries, Wellington, New Zealand.
- Ministry for Primary Industries (MPI) (2016) Review of Management Controls for the South Island Longfin and Shortfin Eel Fisheries (LFE 11–16 & SFE 11–16) in 2016. *MPI Discussion Paper, No: 2016/15*: 33.
- Ministry for the Environment (2008). *Climate Change Effects and Impacts Assessment: A Guidance Manual for Local Government in New Zealand*. 2nd Edition. Mullan, B., Wratt, D., Dean, S., Hollis, M., Allan, S., Williams, T., Kenny, G., and MfE. Ministry for the Environment, Wellington: 149.
- Najera-Hillman, E., Alfaro, A.C., Breen, B.B., O'Shea, S. (2009) Characterisation ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ isotopes) of the food webs in a New Zealand stream in the Waitakere Ranges, with emphasis on the trophic level of the endemic frog *Leiopelma hochstetteri*. *New Zealand Journal of Zoology*, 36: 165–176.
- Olsen, D.A., Tremblay, L., Clapcott, J., Holmes, R. (2012) Water temperature criteria for native aquatic biota. *Auckland Council Technical Report 2012/036*. October 2012.
- Pike, C., Crook, V., Gollock, M. (2019) *Anguilla australis*. The IUCN Red List of Threatened Species 2019: e.T195502A2382653. <https://dx.doi.org/10.2305/IUCN.UK.2019-2.RLTS.T195502A2382653.en>. Downloaded on 27 September 2019.

- Riemann, L., Alfredsson, H., Hansen, M.M., Als, T.D., Nielsen, T.G., Munk, P., Aarestrup, K., Maes, G.E., Sparholt, H., Petersen, M.I., Bachler, M., Castonguay, M. (2010) Qualitative assessment of the diet of European eel larvae in the Sargasso Sea resolved by DNA barcoding. *Biology Letters*, 6: 819–822.
- Rowe, D.K. (1999) Factors influencing the abundance of the common bully, *Gobiomorphus cotidianus* McDowall, in small, North Island, New Zealand, lakes. *Fisheries Management and Ecology*, 6: 377–386.
- Ruru, I., Chisnall, B.L. (2007) Management of customary eel fisheries - The Waipaoa River catchment. *Final research report for the Ministry of Fisheries: EEL2005-02*. Maumahara Consultancy Services Ltd, Gisborne, New Zealand: 61.
- Ryan, P.A. (1986) Seasonal and size-related changes in the food of the short-finned eel, *Anguilla australis* in Lake Ellesmere, Canterbury, New Zealand. *Environ Biology Fish*, 15: 47–58.
- Sagar, P.M., Glova, G.J. (1998) Diel feeding and prey selection of three size classes of shortfinned eel (*Anguilla australis*) in New Zealand. *Marine and Freshwater Research*, 49: 421–428. DOI: <https://doi.org/10.1071/MF97154>.
- Shiao, J.C., Tzeng, W.N., Collins, A., Jellyman, D.J. (2001) Dispersal pattern of glass eel stage of *Anguilla australis* revealed by otolith growth increments. *Marine Ecology Progress Series*, 219: 241–250.
- Stuart, R.E., Closs, G.P., Lokman, P.M., Jellyman, D. (2019) The influence of environmental cues on the activity of silver eels (*Anguilla dieffenbachii*) in Lake Manapouri, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 53: 169–181, DOI: 10.1080/00288330.2018.1483955
- Todd, P.R. (1981) Morphometric changes, gonad histology, and fecundity estimates in migrating New Zealand freshwater eels (*Anguilla* spp.). *New Zealand Journal of Marine and Freshwater Research*, 15: 155–170. DOI: 10.1080/00288330.1981.9515908
- Townsend, A.J., de Lange, P.J., Duffy, C.A.J., Miskelly, C.M., Molloy, J., Norton, D.A. (2007) New Zealand Threat Classification System manual. Department of Conservation, Wellington. 35 p.
- Waikato Raupatu River Trust (2014) *Waikato River Fisheries Bylaws for the Waikato Tainui Fisheries Area*. Waikato Raupatu River Trust, Hamilton.
- Watanabe, S., Aoyama, J., Tsukamoto, K. (2006) Confirmation of morphological differences between *Anguilla australis australis* and *A. australis schmidtii*. *New Zealand Journal of Marine and Freshwater Research*, 40: 325–331.
- Watene, E., Boubée, J., Haro, A., Roper, D. (2003) Downstream Movement of mature eels in a hydro-electric reservoir in New Zealand. In: *Biology, Management, and Protection of Catadromous Eels*. Dixon, D. (Ed.). *American Fisheries Society Symposium*, 33: 191–205.
- Williams, E.K, Boubée, J.A.T., Paterson, C. (2011) Assessment of the eel fishery in the Kawakawa (Taumārere) River catchment. *New Zealand Fisheries Assessment Report 2011/28*. June 2011: 51.

Williams, E., Crow, S., Murchie, A., Tipa, G., Egan, E., Kitson J., Clearwater, S., Fenwick, M.
(2017) Understanding Taonga Freshwater Fish Populations in Aotearoa-New Zealand.
NIWA Client Report No. 2017326HN. NIWA Wellington.

8 Tuna / Longfin eel (*Anguilla dieffenbachii*)

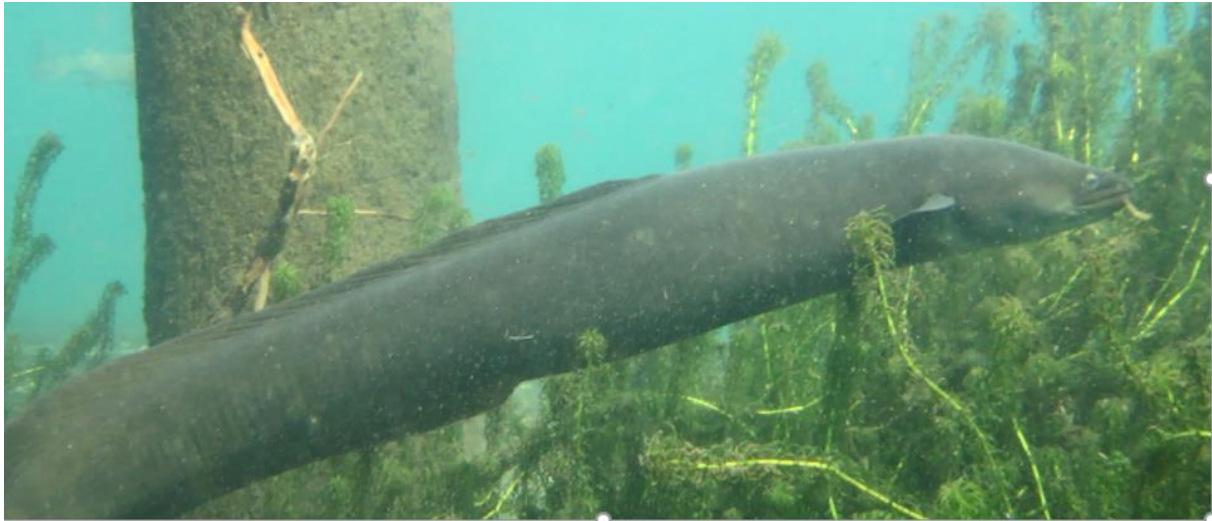


Figure 8-1: Adult longfin eel (*Anguilla dieffenbachii*). Source: NIWA.

8.1 Habitats

Longfin eels (Figure 8-1) are found in numerous different habitat types (n=7) over their life cycle (see Supplementary Appendix 2). Longfins are found in two marine neritic (pelagic habitats and estuaries), two marine/oceanic (Epipelagic [0-200 m] and Mesopelagic [200-1,000 m]) (see Figure 7-2 in section 7) three wetland habitat types (Permanent Rivers/Streams/Creeks [includes waterfalls]; Permanent Freshwater Lakes (over 8 ha) and Permanent Freshwater Marshes/Pools (under 8 ha).

Juvenile glass eels (Figure 8-2) are found in coastal waters, estuaries and the lower reaches of rivers. They typically migrate between July and December. Longfin glass eels appear indifferent to the freshwater type they choose to enter, a response in keeping with their broader habitat preferences (McCleave and Jellyman 2002).



Figure 8-2: A longfin glass eel. Source: NIWA.

Adult longfins prefer flowing water and hence are found extensively in mainstem rivers. They penetrate long distances inland and inhabit high country lakes and rivers. Although juveniles of both species prefer shallow water and coarse substrata, adults prefer deep, slow-moving water. Larger eels (more than 300 mm) of both species are commonly associated with cover, such as macrophyte beds, willow roots, overhanging banks, in-stream debris and shade. As eels increase in size the preference is for deeper water, where eels over 499 mm generally prefer water almost twice as deep. Once longfin eels find a suitable habitat, movement may be limited, with an average home range of 10 m (Jellyman and Sykes 2003).

In an analysis of the microhabitat preferences of longfins in tributaries, variables strongly associated with longfin eel capture were maximum depth (longfin eels used microhabitat of intermediate depth), distance to pool (intermediate), sediment size (finer), sediment depth (deeper), stream width (intermediate) and flow velocity (slower; Broad et al. 2001). Winter longfin eel locations were characterized by deeper and finer sediments in contrast to summer and spring locations (Broad et al. 2001).

A recent study suggests that roughly one-third of the available longfin habitat is currently commercially fished (Beentjes et al. 2016) and approximately 40% of the longfin eel habitat is estimated to be impacted by both hydroelectric dams and commercial fishing.

8.2 The fishery

Information on the commercial freshwater eel fishery was derived from the executive summaries in Beentjes (2019) and are reported for the North Island and South Island fisheries. Routine Catch-Per-Unit Effort (CPUE) analyses are used to routinely monitor longfin eel stocks in Aotearoa-New Zealand. Fisheries New Zealand (2018) reports landings (t) of 'longfin' from 1989–90 to 2015–16. Stock assessment modelling for the longfin fishery in Southland and subsequently Aotearoa-New Zealand-wide was attempted (Dunn et al. 2009; Fu et al. 2012); however, the Ministry for Primary Industries (MPI) rejected the stock assessment models because of the underlying assumptions on estimates of longfin commercial catches and recruitment from individual eel statistical areas (Beentjes 2019).

The commercial tuna fishery is currently managed as 11 separate longfin and shortfin stocks (see Figure 7-4 in section 7) with the 2016–17 TACC set at 163 tonnes and 580 tonnes, respectively (Clements and Associates 2016; MPI 2016). New Zealand's commercial freshwater eel fishery supplies both a domestic and export market and was estimated to have an export value of NZ\$6.1 million by MFish in 2009 (MFish 2009). The asset value of the longfin eel commercial fish resource caught under the QMS (year ended September 2018) was NZ\$13,500,000¹⁷. Since the implementation of the QMS the highest longfin eel total catches were observed in 2005–06 when 129 tonnes were landed (Figure 8-3).

Historic eel landings data from processors dating back to the 1970s indicate that longfin catches have declined relative to shortfin, and the fishery has also been subject to a greater reduction in mean size throughout Aotearoa-New Zealand. The reasons for the decline in longfin catch over time are unknown but are likely to be related to market demand (Beentjes 2019).

¹⁷ <https://figure.nz/chart/LM7iOMLt6SEQscZR>

Year ended September 2004–2018, tonnes

Provider: Stats NZ



Figure 8-3: Total commercial catch (blue line) of longfin eel in Aotearoa-New Zealand between 2003–04 and 2017–18. Source: Figure.NZ and StatsNZ (<https://figure.nz/chart/XuZ4uvsOpxD1Tz8E-Taw2ZuaV2ARStqvl>).

8.2.1 North Island longfin fishery

The commercial freshwater eel monitoring programme began in the North Island in 2003–04. Longfin in the North Island were caught from nearly all subareas (63 of the possible 65) over 15 years, but more than one-third of the catch originated from just four subareas, AA4 (Dargaville), AD10 (Waipā River), AD12 (Lake Waikare, Port Waikato), and AL1 (Lake Wairarapa). The most important subarea (AA4, Dargaville) contributed 107 tonnes or 12% of North Island longfin catch over the 15 years. North Island commercial longfin landed catches over the 15-year period have fluctuated more than shortfin and are characterised by particularly low catches between 2008–09 to 2010–11, and between 2014–15 and 2017–18, with an overall trend of declining catch. Factors that may have influenced annual longfin catches and size ranges include the 58% TACC reductions for North Island longfin stocks for the 2007–08 fishing year, fluctuating market demands, annual rainfall, and more recently, a progressive decline in the availability of ACE (Annual Catch Entitlement) to fishers. The North Island longfin fishery is more prone to fluctuating market demand than shortfin because it is a less marketable species of eel. As for shortfin there has been a reduction and contraction in fishing effort (from 49 subareas per year, on average for the first six years, to 41 subareas per year on average for the last six years) (Beentjes 2019).

North Island commercial longfin catches over the fifteen-year period from 2003–04 to 2017–18 have fluctuated more than six-fold and are characterised by an overall trend of declining landed catch. The medium longfin weight grade is poorly represented in the first three years of low catches (2008–09 to 2010–11).

Some iwi-owned fisheries entities have placed a moratorium on the harvest of longfins, ‘shelving’ (not fishing) their quota, as they are concerned about declining stocks (Kawe 2014; Te Kupenga o Maniapoto Limited 2013). For example, the Maniapoto Māori Trust Board ‘...was concerned that the sustainability of this iconic species was under threat due to overfishing’ (Te Kupenga o Maniapoto Limited 2013). In making their decision, they considered that ‘tuna has a much higher cultural value to our people, far in excess of the earnings derived by the company from ACE (Annual Catch Entitlement) sales’ (Te Kupenga o Maniapoto Limited 2013).

8.2.2 South Island longfin fishery

In the South Island, eel processor data were provided initially in 2006–07 for the Quota Management area (QMA) ANG 15 (Southland/Otago), but no details on specific catch locations were recorded. Provision of more useful data, similar to the North Island, has been collected for the South Island since the 2010–11 fishing year.

Longfin in the South Island were caught from most subareas (53 of the possible 58) between 2010–11 and 2017–18 and although less aggregated than shortfin, half of the catch originated from just seven subareas: AW11 (Mataura River coast), AW9 (Oreti River coast), AW3 (Oreti River inland down to Bog Burn), AV10 (Clutha River coast), AX3 (Grey River Arnold River), AU5 (Waitaki River), and AP2 (Wairau River). The three Southland subareas from the Mataura and Oreti Rivers (AW11, AW9 and AW3) have contributed 236 tonnes or 30% of South Island longfin catch over eight years. The commercial longfin catch in ANG 15 over a 12 year period is highly variable and shows no consistent trend in annual landed catch, although there are indications of a decline in the largest weight grade in recent years.

For the South Island overall, there is a trend of declining longfin landed catch and in the largest weight grade over the eight-year time series. The lower landed catch in recent years can be attributed to lower port price for large longfin, and primarily the split into separate shortfin and longfin stocks in 2016–17. The longfin landed catch is also well below the current TACC introduced in 2016–17 as a result of fisher retirements, withheld quota, and ACE imbalances resulting from the nominal 1 tonne TACCs set in LFE 11 to LFE 14. The longfin catch was stable in the key subareas, but for subareas within LFE 11 to LFE 14 (Nelson, Marlborough and north east of Otago) the pattern has changed dramatically in the last two years as the longfin quota (1 t) is now too low to make it economic or practical to fish these areas (Beentjes 2019).

8.3 Life cycle, age and growth

The life cycle of longfin eels is similar to shortfin eels and they are also catadromous (see Figure 7-6 in section 7). Most of their life is spent in fresh water, but they migrate to the marine environment for reproduction.

Eels have sex-specific life-history strategies. Males may grow faster than females initially, but this difference is soon reversed, and females attain a greater age- and size-at-metamorphosis than males (Davey and Jellyman 2005). The sex of developing gonads is labile, and gender is thought to be determined principally by environmental factors. Individuals experiencing rapid growth prior to gonad differentiation tend to develop as males, whereas eels that grow slowly initially are more likely to develop as females (Davey and Jellyman 2005). High eel densities, temperatures and saline conditions are thought to favour development as males, but experimental studies have failed to demonstrate a clear effect on sex determination (Davey and Jellyman 2005). High proportions of female silver eels migrating from some upstream areas, lakes and large rivers may be due to low population density or poor conditions for growth in these habitats.

Prior to migration, silver eels aggregate in the lower reaches of rivers or at lake outlets waiting for suitable conditions for migration (Jellyman and Unwin 2019). Longfin males generally migrate to sea during April, and longfin females during late April and May. The marine migration of adults to the spawning grounds takes approximately 6-9 months. The migrations of adult eels to the spawning grounds are not well known, but their spawning grounds are believed to be somewhere in the westward flowing -South Equatorial Current (Jellyman and Bowen 2009; Miller et al. 2009; see Figure 7-8 in section 7). The actual location may overlap with that of shortfin eels and lie east of New Caledonia, possibly in the south Fiji basin (Jellyman and Tsukamoto 2005) (Figure 8-4). No longfin eel larvae have ever been found. Larval dispersal modelling studies suggest longfin eel spawning grounds are in the Western Pacific Ocean (Jellyman and Bowen 2009). More recently (January 2020), pop-up satellite tagging has showed mature females heading towards New Caledonia although the tags failed prematurely (Don Jellyman, pers. comm.). Longfin eels are semelparous meaning they die after reproduction.

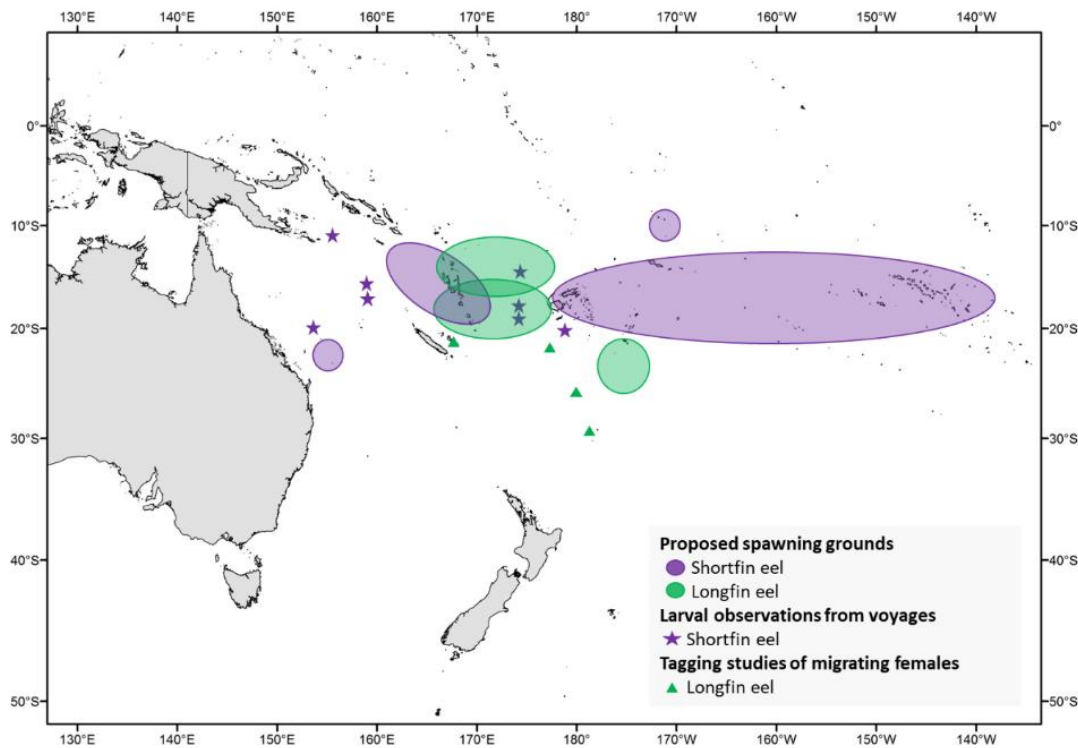


Figure 8-4: Proposed spawning areas for longfin eel in the Western Pacific Ocean. The potential spawning areas of longfin eel are shown in green. Source: NIWA.

Following reproduction, longfin larvae (leptocephalii) traverse more than 20° latitude from their potential spawning grounds in the Western Pacific Ocean to Aotearoa-New Zealand (see Figure 7-8 in section 7). Longfin larvae recruiting to Aotearoa-New Zealand are thought to enter the eastward flowing current of the Tasman Front (TF; Kuroki et al. 2020), although the actual migration route has not been determined.

From analysis of otolith (e.g., Figure 8-5) microstructure, the spawning period of longfin eels extends from August to December indicating winter, spring and summer spawning events. For example, the estimated hatching dates of longfin eels collected at Pigeon Bay Stream on 9 August 1996 were from 15 August to 28 October 1995; those collected from the Arahura River on 22 August 1996 were from 10 September to 10 December 1995; and those collected at the Makara Stream on 4 September 1996

were from 2 to 18 November 1995 (Miho et al. 2001). Longfin glass eels typically enter rivers and streams around Aotearoa-New Zealand between August and December. Regional differences in mean size and condition indicate an arrival pattern from the north in an anti-clockwise dispersal pattern around Aotearoa-New Zealand.



Figure 8-6: Female longfin eel otolith showing the narrow annual rings indicating slow growth rates. This female is from Lake Manapōuri, Southland. Source: NIWA.

Longfins are one of the largest eel species in the world and can attain a size of almost 2,000 mm and more than 50 kg (e.g., Cairns 1941). As with all freshwater eels, females grow to a much larger size than males. Sizes at maturity and key demographics are given in Table 8-1. For longfin eels the range in recorded age at migration is 11–34 years for males, and 27–61 years for females. Longfin eel growth rates are generally slow (Figure 8-6), averaging 2–3 cm/year – with growth rates in the North Island being slightly faster than in the South Island. Consequently, when they reach the minimum commercial size of 220 g, North Island longfins average 14 years of age in fresh water, while their South Island counterparts average 17.5 years (Jellyman 2009). The maximum age of longfin females can exceed 100 years (Jellyman 1995). The smallest eels identified in Aotearoa-New Zealand as male is 300 mm a for longfin, but this can differ between rivers. For example, in the Aparima River, longfin eels differentiated into males mostly at lengths from 300 to 460 mm and ages from 10 to 25+ years (McCleave and Jellyman 2004).

The smallest longfin females identified were 370 mm. In practice, it is usually difficult to confidently identify the sex of most eels until they are about 400 mm for longfins. Males can be as small as 425 mm when they are migratory but are typically 480–740 mm while females are 750–1,800 mm but can be smaller (i.e., 651 mm for Lake Ellesmere; Todd 1980). Sex ratios can be skewed towards males in heavily fished rivers (McCleave and Jellyman 2004). Selective harvest of females can change the structure of a population such that differentiation into males increases. Longevity, delayed sexual maturity, semelparity, and endemism with restricted range make the longfin eel particularly vulnerable to overfishing and associated changes in sex ratios (McCleave and Jellyman 2004).

Table 8-1: Summary of key demographic characteristics for longfin eels.

Key demographic parameters	Estimates	Reference
Maximum size (mm)	2,000	Jellyman (2013)
Minimum size sexual maturity (mm)	480 mm (Male); 750 mm (Female)	Fisheries New Zealand (2018)
Average size at maturity (mm)	602 mm (Male); 1,150 mm (Female)	Jellyman (2013)
Average age at maturity (yrs)	25 (Male); 40 (Female)	Jellyman (2013)
Longevity (yrs)	>100	Jellyman (1995)
Growth parameter (K) ¹⁸	–	–
Fecundity	1–20 million (1,400–1,600 mm length)	Todd (1981)
Larval size _{hatch} (mm)	–	–
Mean egg size (mm)	–	–
Generation length (yrs)	–	–

8.4 Distribution

Longfin eel are a temperate species that are endemic to Aotearoa-New Zealand. They are found in both the North and South Islands as well as Rekohu/Chathams and Stewart offshore Islands (McDowall 1990; Jellyman 2007). Longfin eels can travel up to 300 km inland and are distributed from sea level up to elevations of 1,150 m, allowing individuals to inhabit high country lakes and rivers (McDowall 2000; Jellyman 2007; Figure 8-7). Longfin eels are assumed to be panmictic, with their genetic structure indicating derivation from a single spawning population (Smith et al. 2001; Jellyman 2007; Jellyman 2013).

8.5 Environmental preferences

The specific environmental preferences of longfin **larvae** are not known. Longfin eel larvae have never been found in the Western Pacific Ocean.

The migration of **glass eels** is affected by water temperature (Jellyman 1977; August and Hicks 2008; Jellyman et al. 2009), although the exact nature of this effect is not well understood. August and Hicks (2008) reported that glass eels preferred water temperatures of between 12 and 20°C, with an optimum of 16.5°C, and that temperatures 22°C inhibited migration. Jellyman (2009) reported that glass eel catch was strongly inversely related to water temperature, with 99% of catch in the two years studied occurring at temperatures between 12.6 and 13.1°C.

Adult longfin eels have an upper thermal limit of 37.3°C, at an acclimation temperature of 15°C (Olsen et al. 2012). While longfins eels cannot survive pH less than 4, survival was observed at 100% at pH 4.5 (Jellyman and Harding 2014).

¹⁸ Eel growth rates are linear (Graynoth and Taylor, 2004).

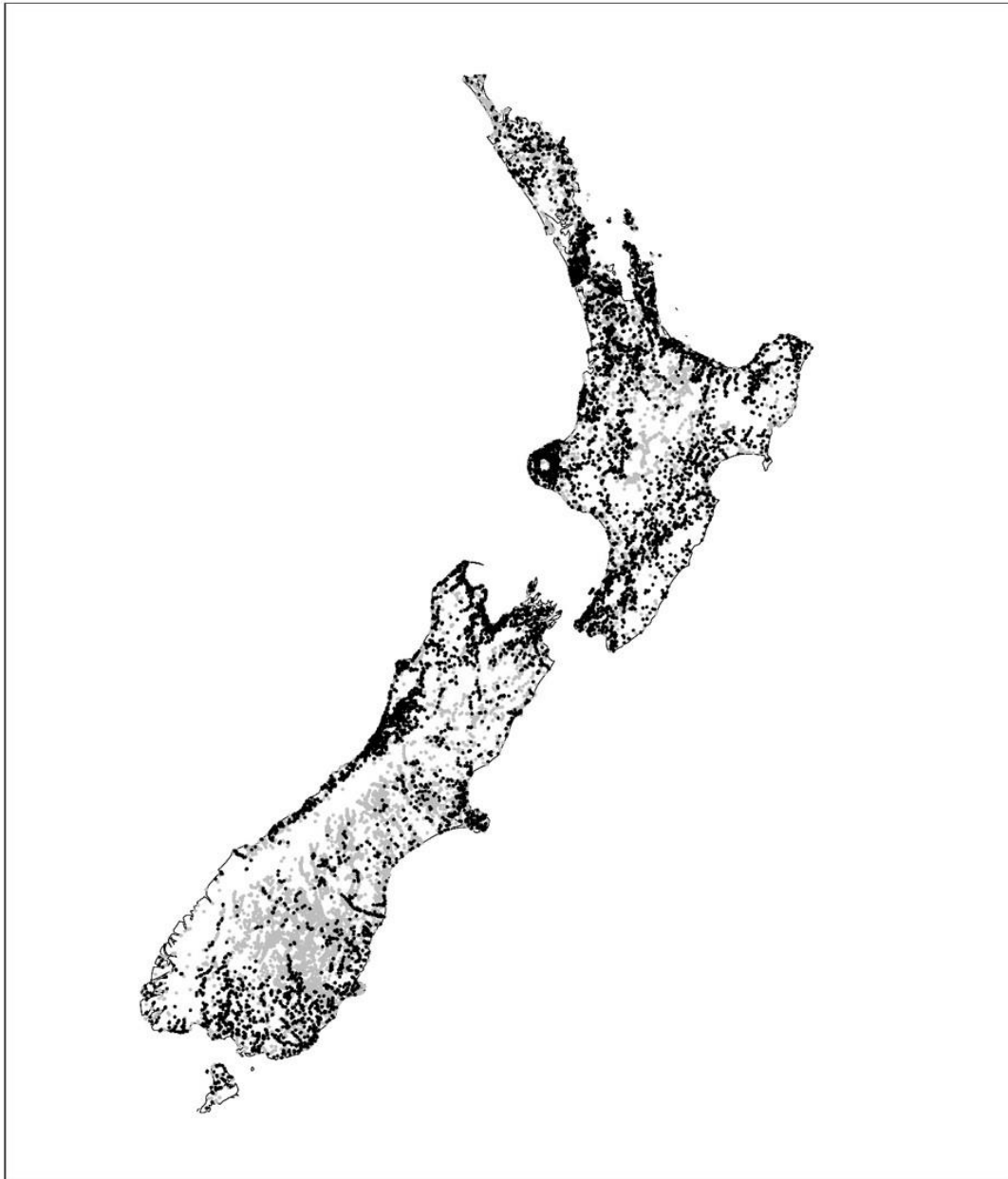


Figure 8-7: Distribution of longfin eel in Aotearoa-New Zealand. Locations of NZFFD records where longfin eel are present (black circles) and absent (grey circles). Source: NIWA.

Environmental variables have been shown to influence **adult** longfin eel movements and migrations. Summer instream temperatures, rain days and seasonality in temperature were the environmental variables correlated with longfin eel distribution (Leathwick et al. 2008; Table A-1). Increases in rainfall and flow have also been suggested as key factors, for example, the emigration of female longfin eels from Lake Manapōuri generally occurs during large outflows (Jellyman and Unwin 2019). Longfins are generally considered less responsive to changing water levels than shortfin eels, because they predominantly occupy habitats in tributaries (Jellyman 2013).

Information on the timing of downstream migrating adults has been successfully used to reduce mortality for a variety of fish species at dams and other passage barriers (e.g., Benstead et al. 1999). In cases where downstream migration can be predicted, implementing mitigation activities such as

targeted netting (Boubée et al. 2001), spillway opening (Watene et al. 2003; Watene and Boubée 2005), or bypass opening (Boubée and Williams 2006) have resulted in reduced injury and mortality rates. However, predicting migration is difficult, especially in rivers whose flow patterns are regulated by storage and generation schedules (Haro et al. 2003). Boubée et al. (2001) found arrival of 60% of the migrant eels at Lake Aniwhenua was attributed to rainfall exceeding a cumulative total of 40 mm over three days. This study also found migrations generally began as water temperatures declined in autumn, and ended when the temperature dropped below 11°C. Such rainfall and flow triggers could be used as predictors to commence mitigation activities that would allow mature eels to proceed uninjured past barriers such as hydro-electric dams (Boubée et al. 2001).

Table 8-2: Environmental cues for the downstream migration of Aotearoa-New Zealand eels. Grey shaded areas indicate these variables were not tested/not significant in the study referenced.

Rainfall/flow	Lunar cycle	Pressure changes	Temperature	Reference
Floods	Not moonlit nights			Downes (1918)
Early stages of floods	Not moonlit nights			Best (1929)
Floods				Cairns (1941)
High rainfall	Low catches on full moon	Important		Burnet (1969)
Rainfall	Low catches on full moon	Important with rainfall and dark moon		Todd (1981)
Floods	No effect	No obvious effect	Less than 11°C inhibiting	Boubée et al. (2001)
Rainfall	New moon best			Watene et al. (2003)
Rainfall	No effect			Boubée and Williams (2006)
High lake outflow	No effect			Jellyman and Unwin (2019) Stuart et al. (2018)

8.6 Diet and predation

The marine diet of longfin eel **larvae** is not known but presumably they feed on marine snow, zooplankton and gelatinous plankton (Riemann et al. 2010; Ayala et al. 2018) (see Figure 7-10 in section 7). Predators of longfin eel larvae are unknown but may include mesopelagic fishes such as lantern fish species.

Longfin eel **adults** are commonly the apex predator in Aotearoa-New Zealand's freshwater food webs. Eels are usually more active at night and rest during the day. Feeding slows down during cold temperatures (less than 10°C). During a flood, eels will move out of the main river channel to the margins where they will feed (Chisnall 1987; Chisnall 2000).

The diet of longfins varies with size (gape of the mouth). Small eels eat aquatic invertebrates – larval mayflies, caddisflies, snails etc. Once longfin reach 400 mm, the adults are piscivorous and feed on a range of fish species (Kelly and Jellyman 2006). In high-country lakes upland bullies (*Gobiomorphus*

breviceps) are important for eels less than 500 mm, but yearling brown trout (*Salmo trutta*) were more important for larger eels (more than 700 mm; Jellyman 1996). In some studies, no seasonal differences in diet are found (Sagar and Eldon 1983). Where shortfin and longfin eels co-exist, they have different feeding preferences (Jellyman 1989). Analyses of longfin eel stomach contents during spring have found high numbers of fresh-run adult lampreys, with a large longfin eel reported to contain 12 adult lampreys (M. Bonnett, NIWA, personal communication).

Predators of longfin eel **larvae** in the marine environment are unknown. Mesopelagic fishes (living between 0–300m deep) especially lanternfish (Myctophidae) are known to predate on the larvae of the European eel (Jensen et al. 2018) and may also predate on longfin eel larvae (Flynn and Marshall 2013). Predators of **glass eels** in coastal and estuarine waters are not known. **Elvers** are eaten by several bird species including the Australasian bittern¹⁹, Hochsetters frogs (Najera-Hillman et al. 2009) and by larger longfin and shortfin eels (e.g., Jellyman 1996; Sagar and Glova 1998).

Introduced fish such as trout, perch, gambusia, rudd, catfish, koi carp, and tench can also impact eel populations through predation, competition for food resources, degrading habitat and reducing biodiversity (Rowe 2004).

8.7 Recruitment

Recruitment of longfin glass eels has been shown to be highly variable within and between years. Glass eel recruitment was monitored between 1995–2006 (between the months of September and October). Data were examined by region, with region being either North or South Island (“island effect”) or East or West Coast (“coast effect”). Conclusions from this analysis of glass eel data (Parliamentary Commissioner for the Environment [PCE] 2013) were that:

- Shortfin glass eels were more abundant than longfins;
- There was no evidence that recruitment of longfins had declined over the period of record;
- Results for “newly arrived” glass eels were virtually identical to “all glass eels”. The exception was that while freshly arrived longfin glass eels showed no differences between East or West Coast densities, the West Coast densities exceeded the East coast densities for all longfin glass eels;
- One caveat is that, relative to the generation times of the eels, the time frame of these data is very short. While it is encouraging that recruitment of longfin showed no decline over the 12 years, it would be premature to conclude that recruitment of longfins is stable. Yearly trends in both species were generally similar, indicating that both seemed to be responding to the same factors in the marine environment;
- In the river with the largest glass eel recruitment, the Waikato River, there is some evidence that present-day runs are smaller than 30 years previously, and the overall proportions, and hence numbers of longfins, has declined substantially.

Numbers of longfins recruiting to the Waitaki and Clutha Rivers in the South Island are especially low while electric fishing samples show there was clear evidence of an overall lack of juvenile longfins at sites in both islands. This was most pronounced for South Island sites (PCE 2013).

¹⁹ <http://nzbirdsonline.org.nz/species/australasian-bittern>

Longfin elver abundance can vary two-fold between some years, but overall there has been no trend (Figure 8-8). Matahina Dam and Karāpiro Dam both show similar annual patterns over the 15 years of continuous data collection. Pātea Dam also has high levels of recruitment variability, but the most recent longfin catch index in 2015–2016 is more than double that of next highest year in 2007–2008. Data across all sites have been used by Martin and Bowman (2016) to conclude that there has been no decline in recruitment over the last 25 years.

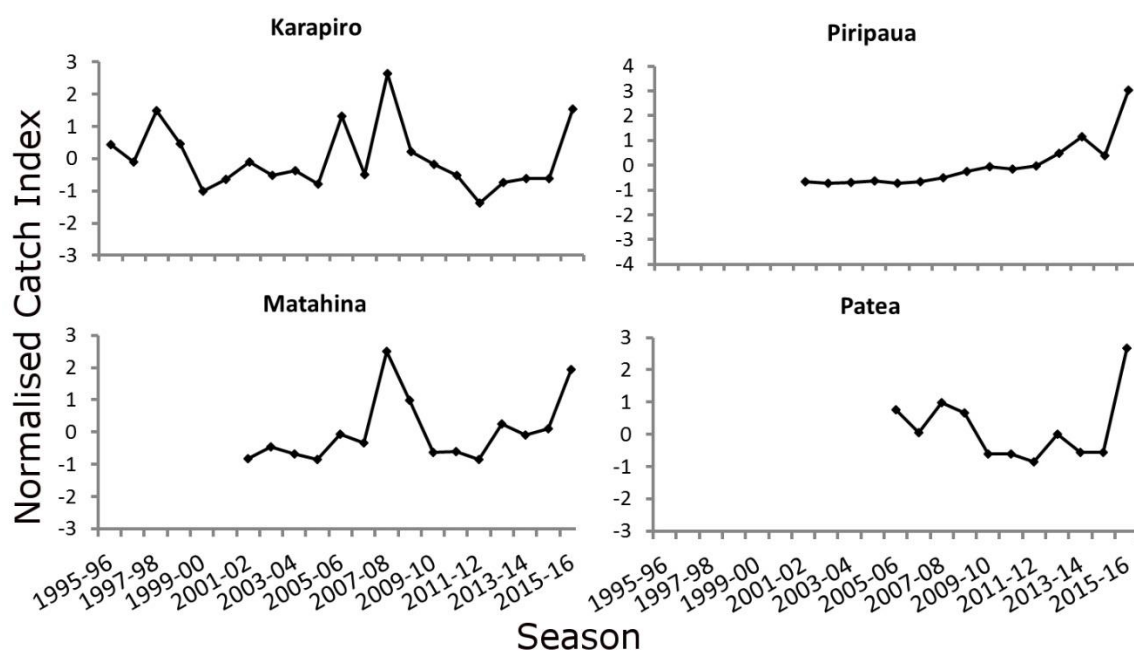


Figure 8-8: Normalised (see Durif et al 2008) index of relative abundance for longfin elver catches from 1995–96 to 2015–16. Source: Martin and Bowman (2016).

8.8 Current impacts of climate change

Droughts are identified as a significant ongoing threat to longfin eels, affecting 50% of the population according to the latest IUCN threat assessment, while temperature extremes are likely to affect longfins in the future (Pike et al. 2019). Stress due to drought may increase mortality rates, while stress associated with temperature extremes may skew sex ratios and reduce reproductive success (Pike et al. 2019).

Between 2016 to 2018, dry summers were associated with reduced commercial catches of longfins in the North Island fishery (Beentjes 2019). More generally, eel catches are greatly influenced by water temperature, flood events (increased catches) and drought conditions (reduced catches). Catches have been observed to decline in winter months (May to September), particularly in the South Island.

8.9 Ecosystem-level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between longfin eel and the wider ecosystem include:

- Longfins often comprise more than 90% of the overall biomass of fish in a given area. As an apex predator, they potentially prey on all other freshwater fish, including introduced species. Longfins control fish populations, including other eels. For instance, when large longfins were removed from a section of stream, smaller eels of both species moved

into the area in considerable numbers (Chisnall et al. 2003) – obviously the presence of the larger eels was exerting a constraint on the distribution of smaller eels;

- Large eels, particularly longfin, play an important role in determining the population structure of eels, including species composition, sex ratios and size distribution. Without this ecological relationship, a higher density of smaller eels can induce sexually immature juveniles to become male. This may have implications not only for interrelated species, but also on the number of female eels contributing to the spawning population²⁰;
- Eels play an important part in recycling nutrients in freshwater ecosystem (Dolamore et al. 2017). Longfin eels contribute marine subsidies to coastal, estuarine and freshwater ecosystems.

8.10 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). The stressors derived from the IUCN threat assessment (n=7), included ‘fishing and harvesting of aquatic resources [large scale harvest]’; ‘dams and water management/use [large and small dams]’; ‘natural system modifications’; ‘agriculture and aquaculture [Small-holder grazing, ranching or farming]’; ‘climate change and severe weather [droughts and temperature extremes]’; and ‘pollution [soil erosion and sedimentation]’ (see Supplementary Appendix 2).

8.11 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality and information gaps took the following into account:

- Allibone et al. (2010) suggests that for species where key aspects of their life histories are unknown (such as longfin eels), it is difficult to understand what is driving any population declines. For longfin eels, we do not know the minimum number of eels that need to reach the spawning grounds to achieve successful spawning and recruitment events (Allibone et al. 2010);
- As summarised in Allibone et al. (2010), for the longfin eel, where concern exists over the survival of females in fresh water (Jellyman 2009), the females that do reach the spawning area must encounter males and spawn. There is an unquantifiable level of risk that if the number of eels in the spawning migration declines to a critical level, then Allee effects will drive reductions in spawning success and juvenile recruitment as eels fail to encounter other eels in the ocean;
- For longfin eel, there exists an additional risk that the population of migrant eels might drop below a minimum threshold required for successful recruitment (Allibone et al. 2010);

²⁰ <https://niwa.co.nz/te-k%C5%ABwaha/tuna-information-resource/biology-and-ecology/maturation-and-identifying-sex>

- Although the most recent elver monitoring data at dam trap and transfer sites throughout Aotearoa-New Zealand indicate recent increases in elver numbers, indicating higher levels of recruitment (Martin and Bowman 2016), this follows a long decline (Jellyman 2009) and it is not apparent if this represents increasing recruitment related to better adult survival, as a result of regulatory changes in the commercial fishery, or is the result of stochastic variations in recruitment.

8.12 References

- Allibone, R., David, B., Hitchmough, R., Jellyman, D., Ling, N., Ravenscroft, P., Waters, J. (2010) Conservation status of New Zealand freshwater fish, 2009. *New Zealand Journal of Marine and Freshwater Research*, 44: 271–287.
- August, S.M., Hicks, B.J. (2008) Water temperature and upstream migration of glass eels in New Zealand: implications of climate change. *Environmental Biology of Fishes*, 81: 195–205.
- Ayala, D.J., Munk, P., Lundgreen, R.B.C., Traving, S.J., Jaspers, C., Jørgensen, T.S., Hansen, K.H., Riemann, L. (2018) Gelatinous plankton is important in the diet of European eel (*Anguilla anguilla*) larvae in the Sargasso Sea. *Scientific Reports*, 8: 6156. DOI: 10.1038/s41598-018-24388-x
- Beentjes, M.P., Sykes, J., Crow, S.K. (2016) GIS mapping of the longfin eel commercial fishery throughout New Zealand and estimates of longfin habitat and proportion fished. *New Zealand Fisheries Assessment Report 2016/32*: 53.
- Beentjes, M.P. (2019) Monitoring commercial eel fisheries: 2015–16 to 2017–18. *New Zealand Fisheries Assessment Report 2019/42*. Fisheries New Zealand, Wellington: 92. <https://fs.fish.govt.nz/Doc/24740/FAR-2019-42-monitoring-commercial-eel-fisheries.pdf.ashx>
- Benstead, J.P., March, J.G., Pringle, C.M., Scatena, F.N. (1999) Effects of a low-head dam and water abstraction on migratory tropical stream biota. *Ecological Applications*, 9: 656–668.
- Best, E. (1929) Fishing methods and devices of the Maori. *Dominion Museum Bulletin*, 12: 1–230.
- Boubée, J.A.T., Williams, E.K. (2006) Downstream passage of silver eels at a small hydroelectric facility. *Fisheries Management and Ecology*, 13: 165–176.
- Boubée, J.A.T., Mitchell, C.P., Chisnall, B.L., Bowman, E.J., Haro, A. (2001) Factors regulating the downstream migration of mature eels (*Anguilla* spp.) at Aniwhenua Dam, Bay of Plenty, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 35: 121–134.
- Broad, T.L., Townsend, C.R., Closs, G.P., Jellyman, D.J. (2001) Microhabitat use by longfin eels in New Zealand streams with contrasting riparian vegetation. *Journal of Fish Biology*, 59: 1385–1400. doi:[10.1111/j.1095-8649.2001.tb00199.x](https://doi.org/10.1111/j.1095-8649.2001.tb00199.x)
- Burnet, A.M.R. (1969) Migrating eels in a Canterbury river, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 3: 230–243.

- Cairns, D. (1941) Life-history of the two species of New Zealand freshwater eel. Part I. Taxonomy, age, growth, migration, and distribution. *New Zealand Journal of Science and Technology*, 23: 53–72.
- Chisnall, B.L. (1987) *Juvenile Eel Biology in the Backwaters of the Waikato River*. MSc thesis, University of Waikato, New Zealand.
- Chisnall, B.L. (2000) Utilisation of flood plains by New Zealand freshwater eels. *NIWA Client Report: CEE01201*. Report prepared for Central Eel Enhancement Co Ltd: 15.
- Chisnall, B.L., Martin, M.L., Hicks, B.J. (2003) Effect of harvest on size, abundance, and production of freshwater eels *Anguilla australis* and *A. dieffenbachii* in a New Zealand stream. In *Biology, Management, and Protection of Catadromous Eels*, pp. 177–189. Ed. by D. A. Dixon. *American Fisheries Society Symposium*, 33: 388.
- Clements & Associates Limited (2016) *The Atlas of Area Codes & TACCs 2016-17*. Clements & Associates Limited Report.
- Davey, A.J.H., Jellyman, D.J. (2005) Sex Determination in Freshwater Eels and Management Options for Manipulation of Sex. *Reviews in Fish Biology and Fisheries*, 15: 37–52. DOI: 10.1007/s11160-005-7431-x.
- Dolamore, B., Puddick, J., Wood, S.A. (2017) Accumulation of nodularin in New Zealand shortfin eel (*Anguilla australis*): potential consequences for human consumption. *New Zealand Journal of Marine and Freshwater Research*, 51: 321–332, DOI: 10.1080/00288330.2016.1236738
- Downes, T.W. (1918) Notes on eels and eel-weirs (tuna and pa-tuna). *Transactions and Proceedings of the New Zealand Institute*, 50: 296–316.
- Dunn, A., Beentjes, M.P., Graynoth, E. (2009) Preliminary investigations into the feasibility of assessment models for New Zealand longfin eels (*Anguilla dieffenbachii*). *New Zealand Fisheries Assessment Report 2009/30*. 42 p. Available online via: http://fs.fish.govt.nz/Doc/21903/09_30_FAR.pdf.ashx
- Fisheries New Zealand. (2018) Fisheries Assessment Plenary. Stock Assessment and Stock Status Volume 1: Introductory Section to Groper. May 2018: 511.
- Flynn, A.J., Marshall, N.J. (2013) Lanternfish (Myctophidae) Zoogeography off Eastern Australia: A Comparison with Physicochemical Biogeography. *PLoS ONE*, 8: e80950. doi:10.1371/journal.pone.0080950
- Foden, W.B., Young, B.E., Akçakaya, H.R., Garcia, R.A., Hoffmann, A.A., Stein, B.A., Thomas, C.D., Wheatley, C.J., Bickford, D., Carr, J.A., Hole, D.G., Martin, T.G., Pacifici, M., Pearce-Higgins, J.W., Platts, P.J., Visconti, P., Watson, J.E.M., Huntley, B. (2019) Climate change vulnerability assessment of species. *WIREs Climate Change*, 10: e551.
- Fu, D., Beentjes, M.P., Dunn, A. (2012) Further investigations into the feasibility of assessment models for New Zealand longfin eels (*Anguilla dieffenbachii*). *Final Research Report for Ministry of Fisheries Project EEL200702*: 77.

- Haro, A., Castro-Santos, T., Whalen, K., Wippelhauser, G., McLaughlin, L. (2003) Simulated effects of hydroelectric project regulation on mortality of American eels. *In: Dixon DA. (Ed.) Biology, Management and Protection of Catadromous Eels. American Fisheries Society, Symposium 33, Bethesda, Maryland.*
- Jellyman, D.J. (1977) Invasion of a New Zealand freshwater stream by glass-eels of two *Anguilla* spp. *New Zealand Journal of Marine and Freshwater Research*, 11: 193–209.
- Jellyman, D.J. (1989) Diet of two species of freshwater eel (*Anguilla* spp.) in Lake Pounui, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 23: 1–10.
- Jellyman, D.J. (1995) Longevity of longfinned eels *Anguilla dieffenbachii* in a New Zealand high country lake. *Ecology of Freshwater Fish*, 4: 106–112.
- Jellyman, D.J. (1996) Diet of longfinned eels, *Anguilla dieffenbachii*, in Lake Rotoiti, Nelson Lakes, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 30: 365–69.
- Jellyman, D.J. (2007) Status of New Zealand fresh-water eel stocks and management initiatives. *ICES Journal of Marine Science*, 64: 1379–1386.
- Jellyman, D.J. (2009) Forty years on-the impact of commercial fishing on stocks of New Zealand freshwater eels (*Anguilla* spp.). Eels at the Edge. *American Fisheries Society Symposium*, 58: 37–56.
- Jellyman, D.J. (2013) The status of longfin eels in New Zealand - an overview of stocks and harvest. *NIWA Client Report: CHC2012-006*. January 2013 (revised): 76.
- Jellyman, D.J., Bowen, M.M. (2009) Modelling larval migration routes and spawning areas of anguillid eels of Australia and New Zealand. *American Fisheries Society Symposium*, 69: 255–274.
- Jellyman, D.J., Sykes, J.R. (2003) Diel and seasonal movements of radio-tagged freshwater eels, *Anguilla* spp., in two New Zealand streams. *Environmental Biology of Fishes*, 66(2): 143–154.
- Jellyman, D.J., Tsukamoto, K. (2005) Swimming depths of offshore migrating longfin eels *Anguilla dieffenbachii*. *Marine Ecology Progress Series*, 286: 261–267.
- Jellyman, D.J., Booker, D.J., Watene, E. (2009) Recruitment of *Anguilla* spp. glass eels in the Waikato River, New Zealand. Evidence of declining migrations? *Journal of Fish Biology*, 74: 2014–2033.
- Jellyman, P.G., Harding, J.S. (2014) Variable survival across low pH gradients in freshwater fish species. *Journal of Fish Biology*, 85: 1746–1752.
- Jellyman, D.J., Unwin, M.J. (2019) Fine-scale swimming movement and behaviour of female silver eels, *Anguilla dieffenbachii*, within a lake affected by hydro-power generation. *Fisheries Management and Ecology*, 26: 57–69.

- Jensen, M.R., Knudsen, S.W., Munk, P., Thomsen, P.F., Moeller, P.R. (2018) Tracing European eel in the diet of mesopelagic fishes from the Sargasso Sea using DNA from fish stomachs. *Marine Biology*, 165: 130.
- Kawe, T.P. (2014) Feasibility study for assessment of customary harvest by Kaitiaki in the Te Tai Hauāuru and Tauranga Moana Regions. New Zealand Fisheries Assessment Report 2014/61. October 2014. Ministry of Primary Industries, Wellington, New Zealand.
- Kelly, D.J., Jellyman, D.J. (2007) Changes in trophic linkages to shortfin eels (*Anguilla australis*) since the collapse of submerged macrophytes in Lake Ellesmere, New Zealand. *Hydrobiologia*, 579: 161–173. <https://doi.org/10.1007/s10750-006-0400-0>
- Kuroki, M., Miller, M.J., Feunteun, M., Sasal, P., Pikinging, T., Han, Y.-S., Faliex, E., Acou, A., Dessier, A., Schabetsberger, R., Watanabe, S., Kawakami, T., Onda, H., Higuchi, T., Takeuchi, A., Shimizu, M., Hewavitharane, C.A., Hagihara, S., Taka, T., Kimura, S., Mochioka, N., Otake, T., Tsukamoto, K. (2020) Distribution of anguillid leptocephali and possible spawning areas in the South Pacific Ocean. *Progress in Oceanography*, 180: 102234. <https://doi.org/10.1016/j.pocean.2019.102234>
- Leathwick, J.R., Elith, J., Chadderton, W.L., Rowe, D., Hastie, T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, 35: 1481–1497.
- Martin, M., Bowman, E. (2016) Recruitment of freshwater eels 1995–2015. *New Zealand Fisheries Assessment Report 2016/46*. August 2016: 106.
- McCleave, J., Jellyman, D. (2002) Discrimination of New Zealand stream waters by glass eels of *Anguilla australis* and *Anguilla dieffenbachii*. *Journal of Fish Biology*, 61: 785–800.
- McDowall, R.M. (1990) *New Zealand freshwater fishes: a natural history and guide*. Auckland, Heinemann-Reed: 553.
- McDowall, R.M. (2000) *The Reed Field Guide to New Zealand Freshwater Fishes*. Reed Publishing (NZ) Limited: 224.
- Miho, M., Takaomi, A., Michael, J.M., J.J., Donald, Katsumi, T. (2001) Comparison of early life history between New Zealand temperate eels and Pacific tropical eels revealed by otolith microstructure and microchemistry. *Marine Ecology Progress Series*, 213: 273–284.
- Miller, M.J., Kimura, S., Friedland, K.D., Knights, B., Kim, H., Jellyman, D.J., Tsukamoto, K. (2009) Review of ocean-atmospheric factors in the Atlantic and Pacific oceans influencing spawning and recruitment of anguillid eels. Pages 231–249 *In Challenges for Diadromous Fishes in a Dynamic Global Environment. American Fisheries Society Symposium*.
- Ministry for Fisheries (MFish) (2009) Fisheries Assessment Plenary. Stock Assessments and Stock Status. Ministry of Fisheries, Wellington, New Zealand.
- Ministry for Primary Industries (MPI) (2016) Review of Management Controls for the South Island Longfin and Shortfin Eel Fisheries (LFE 11–16 & SFE 11–16) in 2016. *MPI Discussion Paper*, No: 2016/15: 33.

- Najera-Hillman, E., Alfaro, A.C., Breen, B.B., O'Shea, S. (2009) Characterisation ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ isotopes) of the food webs in a New Zealand stream in the Waitakere Ranges, with emphasis on the trophic level of the endemic frog *Leiopelma hochstetteri*. *New Zealand Journal of Zoology*, 36: 165–176.
- Olsen, D.A., Tremblay, L., Clapcott, J., Holmes, R. (2012) Water temperature criteria for native aquatic biota. *Auckland Council Technical Report 2012/036*. October 2012.
- Parliamentary Commissioner for the Environment (PCE) (2013) On a pathway to extinction? An investigation into the status and management of the longfin eel. *PCE Report*. April 2013. Wellington, NZ: 95.
- Pike, C., Crook, V., Gollock, M. (2019) *Anguilla dieffenbachii*. The IUCN Red List of Threatened Species 2019: <https://dx.doi.org/10.2305/IUCN.UK.2019-2.RLTS.T197276A2480893.en> Downloaded on 29 February 2020.
- Riemann, L., Alfredsson, H., Hansen, M.M., Als, T.D., Nielsen, T.G., Munk, P., Aarestrup, K., Maes, G.E., Sparholt, H., Petersen, M.I., Bachler, M., Castonguay, M. (2010) Qualitative assessment of the diet of European eel larvae in the Sargasso Sea resolved by DNA barcoding. *Biology Letters*, 6: 819–822.
- Rowe, D. (2004) Exotic fish: a growing problem in New Zealand fresh waters. *Water & Atmosphere*, 12: September 2004.
- Sagar, P.M., Eldon, G.A. (1983) Food and feeding of small fish in the Rakaia River, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 17: 213–226.
- Sagar, P.M., Glova, G.J. (1998) Diel feeding and prey selection of three size classes of shortfinned eel (*Anguilla australis*) in New Zealand. *Marine and Freshwater Research*, 49: 421–428.
- Smith, P.J., Benson, P.G., Stanger, C., Chisnall, B.L., Jellyman, D.J. (2001) Genetic structure of New Zealand eels *Anguilla dieffenbachii* and *A. australis* with allozyme markers. *Ecology of Freshwater Fish*, 10: 132–137.
- Stuart, R.E., Closs, G.P., Lokman, P.M., Jellyman, D. (2019) The influence of environmental cues on the activity of silver eels (*Anguilla dieffenbachii*) in Lake Manapōuri, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 53: 169–181.
- Te Kupenga o Maniapoto Limited (2013) Annual Report for the Year Ended 30 September 2013. Te Kupenga o Maniapoto Limited (Te Kupenga), Te Kuiti.
- Todd, P.R. (1980) Size and age of migrating New Zealand freshwater eels (*Anguilla* spp.), *New Zealand Journal of Marine and Freshwater Research*, 14(3): 283–293.
- Watene, E., Boubée, J., Haro, A., Roper, D. (2003) Downstream movement of mature eels in a hydro-electric reservoir in New Zealand. In: Dixon D. (ed.). *Biology, Management, and Protection of Catadromous Eels*. *American Fisheries Society Symposium*, 33: 191–205.
- Watene, E.M., Boubée, J. (2005) Selective opening of hydroelectric dam spillway gates for downstream migrant eels in New Zealand. *Fisheries Management and Ecology*, 11: 69–75.

9 Kākahi / Kāeo / Freshwater mussels (*Echyridella menziesii*)



Figure 9-1: Adult kākahi (*Echyridella menziesii*; Freshwater mussel). Source: NIWA.

9.1 Habitats

According to the IUCN habitat delineations, *Echyridella menziesii* (Figure 9-1) are found in three habitat types (Permanent Rivers/Streams/Creeks [includes waterfalls]; Permanent Freshwater Lakes (over 8ha.) and Aquaculture Ponds (see Supplementary Appendix 2; Moore 2013).

Worldwide, the habitat requirements of freshwater mussel juveniles are poorly understood but are thought to include well-sorted and well-aerated sediments with high quality pore water and a supply of fine detrital material for food (Yeager et al. 1994; Buddensiek 1995; Fenwick and Clearwater 2018). In lakes, *E. menziesii* is common in the littoral zone, with clean sand and angle of slope being the most important environmental variables associated with mussel density (James 1985).

Sedimentation clogs their habitat and reduces survival of juvenile mussels possibly through a combination of decreased dissolved oxygen concentrations and elevated ammonia concentrations. This is supported by recent research into the rearing of juvenile *E. menziesii* (Clearwater et al. 2017).

9.2 The fishery

There is a daily combined bag limit of 50 that applies to shellfish that do not have specific limits, such as freshwater crayfish, freshwater mussels and freshwater shrimp (MPI undated; Williams et al. 2017). There are no species-specific conservation measures in place for kākahi in Aotearoa-New Zealand.

9.3 Life cycle, age and growth

Kākahi have a unique life cycle (Figure 9-2) that relies on an obligate host-dependent phase for life cycle completion. Briefly, males release their sperm into the water in spring where it is taken in by the females to fertilise their eggs which are held inside a special brood pouch in the gill. The tiny eggs develop into larvae known as glochidia (<0.5 mm long). In spring and summer, the glochidia are released into the water column, possibly when the female senses the presence of a suitable fish host. A recent study found that peak glochidia release occurred in February when average monthly water temperatures were >18.8°C (Hanrahan 2019). They attach themselves to a host fish (including kōaro,

tuna, bullies, banded kōkopu) using a little tooth on their shell edge. The best attachment location is thought to be the gill, but they are often found on the fin tips, lips and skin. The glochidia are parasites on the fish host (Fritts et al. 2013) while they transform completely into a juvenile mussel. After about two or three weeks they drop off the fish, presumably into soft, sandy sediments in lakes and riverbeds to develop further.

Studies of reproductive seasonality of *E. menziesii* (at Lake Taupō) and unpublished data from other locations in the North Island, particularly Lake Karāpiro (Waikato) show that for the last five years, ripe female mussels brooding glochidia have been routinely collected from November to March (Clearwater et al. 2014; Clearwater et al. 2017). Planktonic glochidia were found in Lake Taupō when seasonal water temperature increased from 14°C (November) up to 20°C (Clearwater et al., in press).

E. menziesii longer than 30 mm dominate populations and it is rare to find juvenile mussels (Grimmond 1968; James 1985; Roper and Hickey 1994). Adult freshwater mussels can live a long time and individuals of more than 100 mm length have been recorded in previous studies (Ogilvie 1993; Sorrell et al. 2007). Populations in Lake Waipori had a mean age of 20–25 years old (Grimmond 1968), with some individuals aged at over 50 years. In other locations the oldest mussels were 13 years old (61 mm) in Lake Taupō (James 1985) to 33 years (84 mm) in the Waikato River (Roper and Hickey 1994). The life span of freshwater mussels can be problematic because adult mussels may be present in a lake or river but might not be a viable, self-sustaining population because of low juvenile survival. This would be known as a “geriatric” population at risk of local extinction. There is potential that juvenile mussels occur in a different habitat (upstream) from the adults and undergo a migration into adult habitat as they develop (Phillips 2006). For example, Grimmond (1968) found juvenile mussels near the mouths of inflowing rivers.

The size range of *E. menziesii* was 10–70 mm in Lake Taupō, and 20–90 mm at six sites along the Waikato River with median lengths from 48 mm at Taupō to 65 mm in Lake Maraetai (on the Waikato) (James 1985; Roper and Hickey 1994). The Lake Taupō study suggested that *E. menziesii* is reproductively mature at shell lengths more than 37 mm long.

The few published studies of *E. menziesii* populations from South Island locations suggest that longer shell lengths are more common. For example, *E. menziesii* at Lake Tuakitoto (70 km south of Dunedin) were 30–105 mm length (Ogilvie and Mitchell 1995), at Lake Waihola (35 km south of Dunedin) they were 10–110 mm long, with the majority in the 80–100 mm range (Grimmond 1968) and at Ō Tū Wharekai (Ashburton Lakes, 110 km west of Christchurch) the largest size range was in Lake Roundabout at 16–110 mm length (De Winton et al. 2013).

Shell length does appear to be related to mussel age, but ages estimated from annular rings within the shells are highly variable (Grimmond 1968; James 1985; Roper and Hickey 1994). The longest shell examined by James (1985) from Lake Taupō was 60 mm length and had 12 annular rings. The longest shells examined by Roper and Hickey (1994) from the Waikato River were 80–83 mm long and had approximately 12–33 annular rings. Grimmond (1968) provides the most comprehensive study of *E. menziesii* annular rings and estimated the ages of 81–90, 91–100, and 103–106 mm long mussels at 21–30, 31–50 and 51–55 years old respectively.

Sex ratios may be skewed in some populations. For example, in Lake Horowhenua, the data suggests that there may be a 67:33 ratio of female: male mussels in the lake and that approximately 43% of the female mussels were brooding glochidia with the remainder in an earlier stage of development (Fenwick and Clearwater 2018).

Eggs

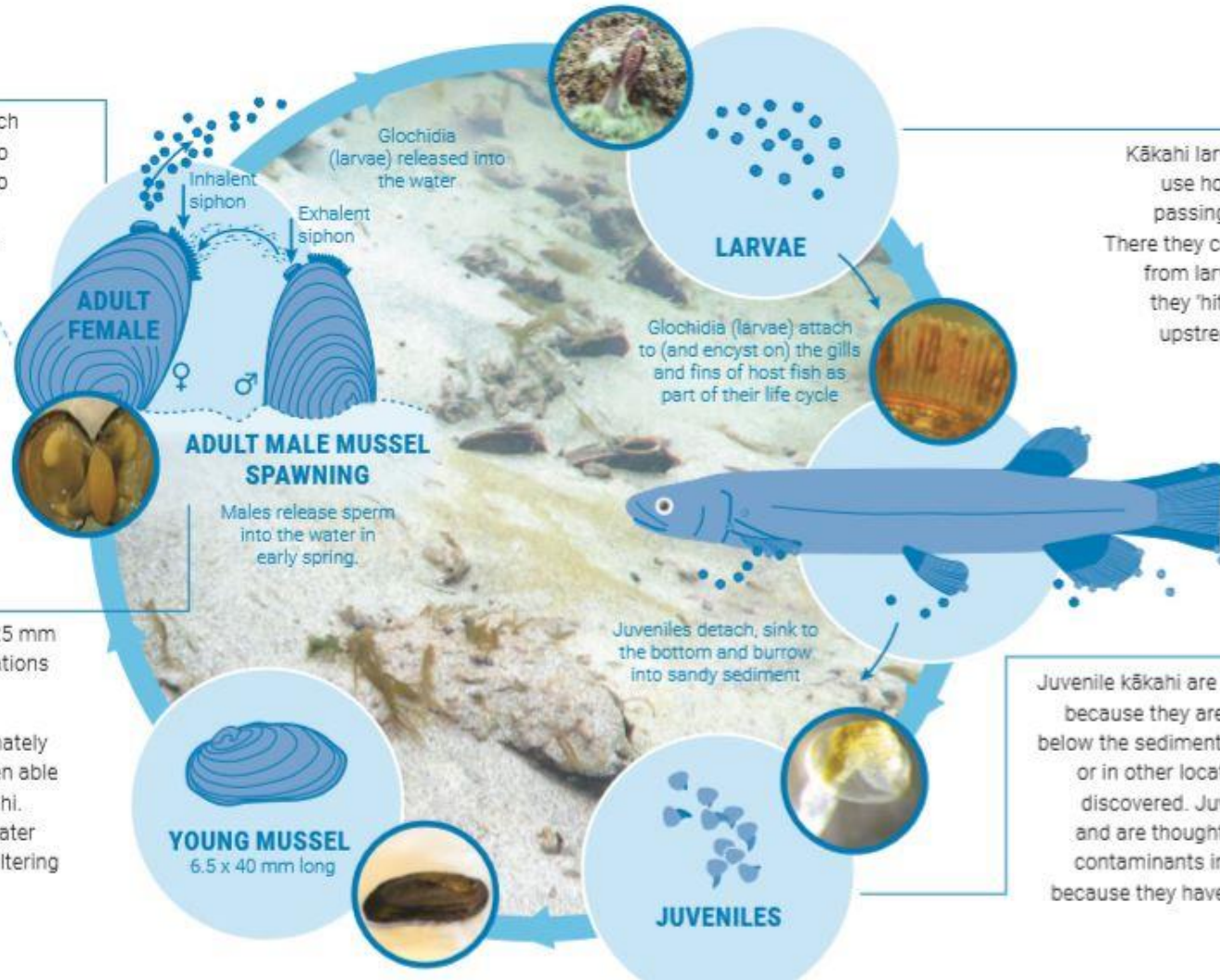
The eggs are fertilised in the gill pouch of the female where they develop into larvae. The female releases them into the water when they are mature – possibly in response to the presence of a fish.



Adult kākahi

Once juveniles have reached about 25 mm length, they move into the same locations as adult mussels.

At about four years of age (approximately 40mm long) they mature and are then able to reproduce with neighbouring kākahi. Males release their sperm into the water and nearby females collect it while filtering to fertilise their eggs.



Larvae

Kākahi larvae called glochidia use hooks to latch on to a passing fishes gills or fins. There they can safely transform from larvae to mussel while they 'hitch a ride', hopefully upstream to good habitat.

Juveniles

Juvenile kākahi are very difficult to find because they are very small and live below the sediment surface in streams or in other locations we haven't yet discovered. Juveniles grow rapidly and are thought to be vulnerable to contaminants in the water possibly because they have a different feeding mode to adults.

Figure 9-2: Freshwater mussel life cycle. Source: NIWA.

Table 9-1: Key demographic parameters of kākahi (*E. menziesii*).

Key demographic parameters	Estimates	References
Maximum size (mm)	>110	de Winton et al. (2013)
Minimum size sexual maturity (mm)	23	Melchior (unpublished data, 2020)
Average size at maturity (mm)	48.8–61.6	Melchior (unpublished data, 2020)
Average age at maturity (yrs)	–	–
Longevity (yrs)	50	Grimmond (1968)
Growth parameter (K)	–	–
Fecundity	28,840–72,000	Melchior et al. (2019)
Larval size _{hatch} (diameter; µm)	277	Melchior et al. (2019)
Mean egg size (mm)	–	–
Generation length (yrs)	–	–

9.4 Distribution and environmental preferences

E. menziesii is widespread throughout Aotearoa-New Zealand and is locally common in some places (Figure 9-3). Research by Fenwick (2006) and Marshall et al. (2014) has provided clear taxonomic designations for Aotearoa-New Zealand’s freshwater mussels but more research is required to confirm their distributions. The juveniles (glochidia) attach to host fish and can be transported upstream to upper catchment areas (Moore 2013) but by and large they are a sessile species as adults. Translocations of *E. menziesii* are occurring to reseed populations (McEwan et al. 2020). *E. menziesii* has been classified as At Risk - Declining with a total area of occupancy >10,000 ha (100 km²) and a predicted decline 10–70% over 3 generations (minimum 10 years) (Grainger et al. 2018).

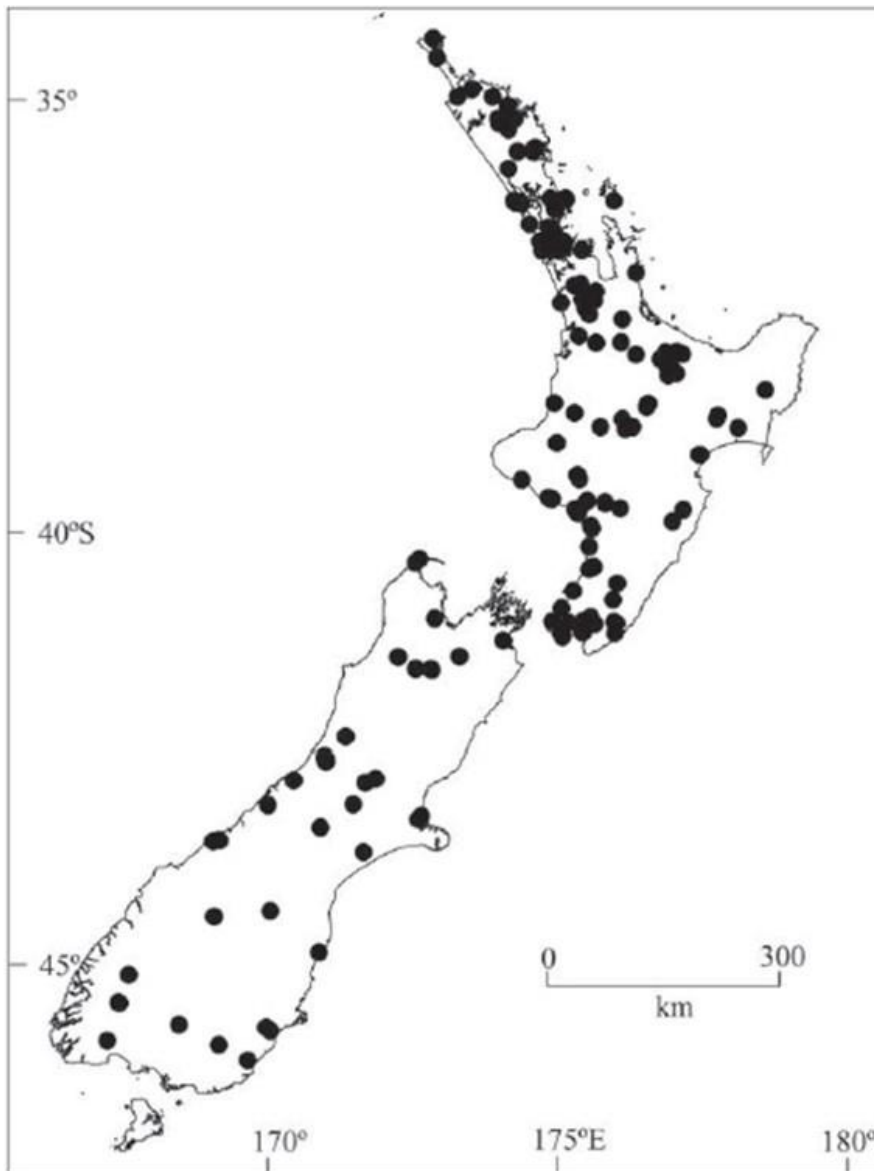


Figure 9-3: Distribution of *Echyridella menziesii*. Source: Marshall et al. (2014).

The habitats and environmental preferences of riverine *E. menziesii* populations are poorly understood. Cyr et al. (2017) has summarised *E. menziesii* distribution in warm monomictic lakes. They found that:

- *E. menziesii* in Aotearoa-New Zealand lakes reach their maximum density within the epilimnion, at increasing depths with increasing lake size and increasing site exposure. The only exceptions were found in large lakes or parts of large lakes with shallow bathymetric slopes, where high mussel densities were found in relatively shallow waters;
- *E. menziesii* density peaks can be found deep in the volcanic Aotearoa-New Zealand lakes;
- *E. menziesii* appear to have a very broad range of depth distribution in these warm monomictic lakes. Mussels were found in deep oligotrophic lakes (down to 12-30 m), often in substantial numbers (up to 2-186 mussels/m²). This depth range is a minimal estimate of the lower boundary of their distribution.

- Mussel distribution in highly productive lakes is limited by hypolimnetic anoxia, and therefore by position of the thermocline;
- *E. menziesii* are found in a wide range of substrates. Mussels were found below the mud deposition boundary in many lakes, suggesting that the presence of fine flocculent organic sediments does not prevent them from living in deep areas; and
- The distribution of *E. menziesii* in warm monomictic lakes appears to be governed by relatively simple physical processes.

Clearwater et al. (2014) demonstrated that *E. menziesii* larvae (glochidia) are extremely sensitive to ammonia exposure when compared to other native invertebrate species. This research is supported by an extensive body of research on the larvae and juveniles of multiple North American freshwater mussel species and strongly suggests that *E. menziesii* juveniles are also.

9.5 Diet and predation

Freshwater mussels are filter-feeders as well as deposit-feeders. They feed on a variety of suspended particulates in the water, including bacteria, phytoplankton, detritus and micro-zooplankton, as well as deposited organic material (e.g., dead plankton, fine silt). Freshwater mussels lack the byssal threads (or “beard”) that some marine mussel species use to attach themselves to substrates; instead they usually partially bury themselves into soft sediments (Williams et al. 2017).

Kōura prey on juvenile mussels (C. Hickey, unpubl. data). It is likely that birds and fish also predate this species, as reported for other mussel species internationally (e.g., Zahner-Meike and Hanson 2001). Rats also predate kākahi, with records of “bitten off” shells from Lake Rototoa, Lake Harihari some Northland lakes, and shell piles/middens observed from Lake Tūtira (Hofstra 2013). Predation of adult freshwater mussels is considered opportunistic and generally occurs when environmental conditions leave mussels exposed (McEwan et al. 2020). In Aotearoa-New Zealand, evidence of rat predation has been observed on freshwater mussel shells on the beach of a hydroelectricity reservoir (Moore et al. 2019). Wading birds feed on kākahi when waters are lowered as part of drainage schemes for farms (McEwan et al. 2020).

9.6 Recruitment

E. menziesii has been extirpated from many lowland shallow waterbodies and is thought to be declining in other lowland lakes throughout the country (Clearwater et al. 2014a; Clearwater et al. 2014b), with potential consequences for benthic-pelagic processes in lakes.

This species is considered functionally extinct in some populations. For example, the smallest mussel collected in Lake Horowhenua was 68 mm long and according to Grimmond (1968) would be approximately 16 years old. In size, and thereby age truncated populations such as Lake Horowhenua, it is apparent the recruitment has failed in the last 15 years (Fenwick and Clearwater 2018).

In 2013 the IUCN ranked *E. menziesii* as being of Least Concern; however, they recognise that there is very little information on juveniles and rates of recruitment, which means it could take a while to notice any decline in populations. The species is thought to be in decline due to reduced recruitment of river populations, and lowland lake populations are also likely to be in decline, however some large upland lake populations may be stable (Moore 2013).

9.7 Current impacts of climate change

A recent study of the micro-geochemistry (as a proxy for climate records) of *E. menziesii* shells from Lake Rotorua supports the contention that dark rings are indeed formed annually, but growth rates decrease markedly as the mussels age (Herath et al. 2017).

9.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between *E. menziesii* and the wider ecosystem include:

- Freshwater mussels are considered key ecosystem engineers due to their highly efficient filter feeding and bioturbating abilities, which enhance water clarity and promote nutrient cycling. Ogilvie and Mitchell (1995) estimated that freshwater mussels were capable of filtering the entire volume of a shallow eutrophic Aotearoa-New Zealand lake every 32 hours, reducing Chlorophyll *a* concentrations. Water quality impacts are likely less pronounced in deeper lakes, as demonstrated by James (1987) who estimated that it would take 6 months for mussels to filter over half of the epilimnetic water volume of oligotrophic Lake Rotokawau (maximum depth 74 m) in the central North Island;
- *E. menziesii* and *E. aucklandica* often occur together (in other words, side-by-side in the sediment) which may indicate that their ecological relationship is more complex than is currently understood. For example, recent surveys have shown the two species, *E. menziesii* and *E. aucklandica*, are usually found within one location (e.g., Lake Wairarapa, Lake Hauroko, west coast Waikato streams) (Greater Wellington Regional Council 2015; Marshall et al. 2014; Hamer et al. 2015);
- Freshwater mussels have a unique relationship with a midge (or chironomid, *Xenochironomus canterburyensis*; Figure 9-4) because the larvae develop within the layers of the freshwater mussel shell (Forsyth 1983). The midge larvae can sometimes be seen in “blisters” on the inside of the shell. Roper and Hickey (1994) found that dead chironomids can become embedded and result in severe shell deformities and flaking.



Figure 9-4: The chironomid, *Xenochironomous canterburyensis* found in freshwater mussel shells. Source: Mark Fenwick.

9.9 Additional (multiple) stressors

Climate change is likely to interact with a range of existing pressures, exacerbating their effects (see references in Foden et al. 2019). The stressors derived from the IUCN threat assessment (n=5), included ‘dams and water management/use [dams]’; ‘pollution [soil erosion/sedimentation, sewage, run-off]’ and ‘transportation and service corridors [shipping lanes]’ (see Supplementary Appendix 2).

9.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- Shell morphology varies with water quality, flow, and wave action, making it difficult in some locations to tell freshwater mussel species apart (Phillips 2006). More work is needed to resolve freshwater mussel taxonomy in Aotearoa-New Zealand;
- More research into the translocation of *E. menziesii* to increase or reseed populations is needed. *E. menziesii* larvae can be extracted from ripe female mussels, transformed in vitro without a host fish, and then grown out with a relatively high survival rate until they are approximately two months old (Clearwater et al. 2017). Survival declines markedly after this point but research into improving juvenile mussel grow-out is continuing and may soon offer an opportunity to maintain a captive or translocated population of the mussels such that they can be used to restore populations.

Host fish are thought to provide transport of mussel larvae to favourable habitat. If tributaries to Lake Horowhenua have higher water quality (than the lake), suitable sediments, and are accessible to host fish species, they may provide both juvenile and adult mussel habitat – and a means to extend the long-term survival of sub-population of *E. menziesii*, until lake restoration efforts take effect. The goals

of New Zealand National Freshwater Mussel Conservation Strategy are outline in Table 9-2 and seek to fill in key knowledge gaps.

Table 9-2. Goals of the New Zealand National Freshwater Mussel Conservation Strategy, 9 September 2015.
(Adapted from Haag and Williams 2014).

Goal No.	Description
1	Work together, increase cooperation and communication amongst entities that study, manage, conserve or restore freshwater mussels.
2	Increase knowledge of mussel population status and trends, including: <ul style="list-style-type: none"> a. Develop basic monitoring protocols (e.g., community groups); b. Develop detailed monitoring protocols for use nationally; c. Include traditional and local knowledge; d. Provide website for both presence/absence records, and data collection tailored to national protocols; e. Provide website on mussel information, updates (e.g., quarterly) on this process and contacts.
3	Grow knowledge of their biology (especially reproduction and host requirements) and habitat requirements (especially for juveniles).
4	Protect and reverse the decline of quality mussel habitat
5	Determine what are the key mechanisms of mussel decline in shallow lakes; what are the mechanisms of their probable decline in streams? Develop restoration guidance from the findings of this research.
6	Enhance public and government understanding and support for freshwater mussel conservation and habitat protection [e.g., flagship species for water conservation?].
7	Develop and trial techniques for holding and translocating large numbers of adult mussels.
8	Develop and trial techniques for reseeding juveniles on a large scale.
9	Increase available funding levels and develop other means to increase mussel conservation efforts.

9.11 References

- Buddensiek, V. (1995) The culture of juvenile freshwater pearl mussels *Margaritifera margaritifera* L. in cages: A contribution to conservation programmes and the knowledge of habitat requirements. *Biological Conservation*, 74: 33–40.
- Butterworth, J. (2008) Lake Rotokākahi: The kākahi (*Hyridella menziesii*) in a general framework of lake health. MSc thesis, University of Waikato, Hamilton.
- Crow, S.K., Snelder, T.H., Jellyman, P.G., Greenwood, M.J., Booker, D.J., Dunn, A. (2016) Temporal trends in the relative abundance of New Zealand freshwater fishes: analysis of the New Zealand Freshwater Fish Database records. *NIWA Client Report CHC2016-049*: 71.
- Clearwater, S.J., Roper, D.S., Hickey, C.W., Martin, M.L., Williams, E.K. (submitted). The reproductive cycle of the freshwater mussel *Echyridella menziesii* (Unionacea: Hyriidae) (Gray, 1843) in Lake Taupo, New Zealand. Submitted to *Hydrobiologia*.
- Clearwater, S.J., Wood, S.A., Ginkel, R.V., Phillips, N.R., Parkyn, S.M. (2012) Toxicity thresholds for juvenile freshwater mussels *Echyridella menziesii* and crayfish *Paranephrops planifrons*, after acute cell-free, or chronic dietary exposure to microcystin-containing crude extracts from *Microcystis* sp. *Environmental Toxicology*, 29: 487–502.
- Clearwater, S.J., Thompson, K.J., Hickey, C.W. (2014) Acute toxicity of copper, zinc and ammonia to larvae (glochidia) of a native freshwater mussel *Echyridella menziesii* in New Zealand. *Archives of Environmental Contamination and Toxicology*, 66: 213–226.
- Collier, K.J., Clearwater, S.J., Neijenhuis, P., Wood, S.A. (2017) Factors influencing biodeposit production by the New Zealand freshwater mussel *Echyridella menziesii*. *New Zealand Journal of Marine and Freshwater Research*, 52: 1–15.
- Cyr, H., Collier, K.J., Clearwater, S.J., Hicks, B.J., Stewart, S.D. (2016) Feeding and nutrient excretion of the New Zealand freshwater mussel *Echyridella menziesii* (Hyriidae, Unionida): Implications for nearshore nutrient budgets in lakes and reservoirs. *Aquatic Sciences*, 79: 557.
- Cyr, H., Phillips, N., Butterworth, J. (2017) Depth distribution of the native freshwater mussel (*Echyridella menziesii*) in warm monomictic lakes: Towards a general model for mussels in lakes. *Freshwater Biology*, 62: 1487–1498.
- de Winton, M., Sutherland, D., Clayton, J. (2013) Kākahi (freshwater mussel) survey of the Ō Tū Wharekai Lakes. *NIWA Client Report No: HAM2013-001*: 26.
- Fenwick, M.C. (2006) *The molecular phylogenetics of the New Zealand freshwater mussels*. Unpublished MSc thesis, Victoria University, Wellington.
- Fenwick, M.C., Clearwater, S.J. (2018) Kākahi survey of Lake Horowhenua. *NIWA Client Report No: 2018181HN*. April 2020. NIWA, Wellington.
- Forsyth, D. (1983) Chapter 9: Animals of the lake-bed. In: DJ Forsyth & C Howard Williams (Eds.). *Lake Taupo: Ecology of a New Zealand lake*. Science Information Publishing Centre, DSIR, Wellington.

- Fritts, M.W., Fritts, A.K., Carleton, S.A., Bringolf, R.B. (2013) Shifts in stable-isotope signatures confirm parasitic relationship of freshwater mussel glochidia attached to host fish. *Journal of Molluscan Studies*, 79: 163–167.
- Grainger, N., Harding, J., Drinan, T., Collier, K., Smith, B., Death, R., Makan, T., Rolfe, J. (2018) Conservation status of New Zealand freshwater invertebrates, 2018. *New Zealand Threat Classification Series 28*. Wellington: Department of Conservation: 25.
- Greater Wellington Regional Council (2015) Kākahi Monitoring Report: Kākahi monitoring for the Wairarapa Moana Wetlands Project. March 2015: 8.
- Grimmond, N.M. (1968) Observations on growth and age in *Hyridella menziesi* (Mollusca: Bivalvia) in a freshwater tidal lake. Unpublished MSc thesis, University of Otago, Dunedin.
- Haag, W.R., Williams, J.D. (2014) Biodiversity on the brink: an assessment of conservation strategies for North American freshwater mussels. *Hydrobiologia*, 735: 45–60.
- Hamer, M., Pingram, M., Collier, K. (2015) Freshwater mussels in Waikato’s West Coast streams. Presentation to the 2015 Joint Annual Conference for NZFSS and ASL (Australian Society of Limnology). 23-26 November 2015. Silverstream, Wellington.
- Hanrahan, N.J. (2019) Field and laboratory investigations of *Echyridella menziesii* (Unionida: Hyriidae) interactions with host fishes. MSc thesis, University of Waikato, Hamilton, New Zealand. Retrieved from <https://hdl.handle.net/10289/12970>
- Hastie, L.C., Cosgrove, P.J., Ellis, N., Gaywood, M.J. (2003) The threat of climate change to freshwater pearl mussel populations. *AMBIO: A Journal of the Human Environment*, 32: 40–46.
- Herath, D., Stern, R., Jacob, D., Clearwater, S., and Fallon, S. (2017) High resolution seasonal signals in bivalve shells from Lake Rotorua, New Zealand. In: ‘Goldschmidt 2017’, 13–18 August 2017, Paris, France. [Abstract]. Available at <https://goldschmidt.info/2017/abstracts/abstractView?id=2017001380> [Verified 18 September 2018].
- Hofstra, D.E. (2013) Flora and fauna survey report for the *Hydrilla* eradication response 2014. *NIWA Client Report HAM2013-033*. NIWA, Hamilton.
- James, M. (1985) Distribution, biomass and production of the freshwater mussel, *Hyridella menziesi* (Gray), in Lake Taupo, New Zealand. *Freshwater Biology*, 15: 307–314.
- James, M. (1987) Ecology of the freshwater mussel, *Hyridella menziesi* (Gray) in a small oligotrophic lake. *Arch Hydrobiol*, 108: 337–348.
- Marshall, B.A., Fenwick, M.C., Ritchie, P.A. (2014) New Zealand Recent Hyriidae (Mollusca: Bivalvia: Unionida). *Molluscan Research*, 34: 181–200.
- McEwan, A.J., Dobson-Waitere, A.R., Shima, J.S. (2020) Comparing traditional and modern methods of kākahi translocation: implications for ecological restoration. *New Zealand Journal of Marine and Freshwater Research*, 54: 102–114.

- Melchior, M., Collier, K., Clearwater, S. (2019) First record of complex release strategies and morphometry of glochidia in sympatric *Echyridella* species (Bivalvia: Unionida: Hyriidae). *Hydrobiologia*. DOI <https://doi.org/10.1007/s10750-019-03995-3>
- Moore, S. (2013) *Echyridella menziesii*. The IUCN Red List of Threatened Species 2013: e.T198678A2532703. <http://dx.doi.org/10.2305/IUCN.UK.2013-2.RLTS.T198678A2532703.en>. Downloaded on 05 July 2017.
- Ministry for Primary Industries (MPI). (undated). Fisheries Plan Freshwater Species. 28 p. Available online via: <http://fs.fish.govt.nz/Doc/16403/FreshwaterSpeciesDraft%20FishPlan%20-%20Current%20State.pdf.ashx>
- Ogilvie, S. (1993) The effects of the freshwater mussel *Hyridella menziesi* on the phytoplankton of a shallow Otago lake. Unpublished MSc thesis, University of Otago, Dunedin.
- Ogilvie, S., Mitchell, S. (1995) A model of mussel filtration in a shallow New Zealand lake, with reference to eutrophication control. *Archives of Hydrobiology*, 133: 471–481.
- Phillips, N. (2006) Review of the potential for biomanipulation of phytoplankton abundance by freshwater mussels (kākahi) in the Rotorua lakes. *NIWA Client Report HAM2006-125*. September 2006: 22.
- Roper, D., Hickey, C. (1994) Population structure, shell morphology, age and condition of freshwater mussel *Hyridella menziesi* (Unionacea: Hyriidae) from seven lake and river sites in the Waikato River system. *Hydrobiologia*, 284: 205–217.
- Sorrell, B., Phillips, N., Wells, R., Sykes, J. (2007) Lake Matiri assessment. *NIWA Client Report CHC2007-089*. June 2007: 49.
- Williams, E., Crow, S., Murchie, A., Tipa, G., Egan, E., Kitson J., Clearwater, S., Fenwick, M. (2017) Understanding Taonga Freshwater Fish Populations in Aotearoa-New Zealand. *NIWA Client Report No. 2017326HN*. NIWA Wellington.
- Yeager, M.M., Cherry, D.S., Neves, R.J. (1994) Feeding and Burrowing Behaviours of Juvenile Rainbow Mussels, *Villosa iris* (Bivalvia:Unionidae). *Journal of the North American Benthological Society*, 13: 217–222.
- Zahner-Meike, E., Hanson, J.M. (2001) Effect of muskrat predation on naiads. In: Bauer G, Wachtler K. (Eds) *Ecology and evolution of the freshwater mussels Unionoida*. Ecological Studies. Springer, Berlin Heidelberg, Germany: 163–183.

10 Kōura / Kēwai / Freshwater crayfish (*Paranephrops planifrons*)



Figure 10-1: Adult kōura (*Paranephrops planifrons*). Source: NIWA.

10.1 Habitats

Kōura (*Paranephrops planifrons*; Figure 10-1) are found in two habitat types according to the IUCN habitat delineations: permanent rivers and streams/lakes greater than eight hectares (see Supplementary Appendix 2). However, it is recognised that *P. planifrons* can occur throughout a much wider range of aquatic habitats including lakes, streams and rivers, as well as reservoirs, ponds, swamps and drains (Chapman et al. 2011). Kōura are rare in low calcium waters (i.e., Lake Tikitapu).

Kōura use low flow refugia amongst stream substrates and cover at flow velocities of up to 1.6 m s^{-1} , with stream edge cover being particularly important as refugia for young of the year kōura (Jowett et al. 2008). The microhabitats of juvenile kōura (less than 8 mm orbit carapace length) include substrate size, and the area of bank undercuts and organic cover types and root mat area (Parata 2019). Preferred microhabitats of adults are associated with cover. They can be found under tree roots, leaf litter, undercut banks, and fallen logs/woody debris, particularly where this material from riparian sources is associated with banks alongside pools or forms debris clusters (Parkyn and Collier, 2004; Jowett et al. 2008). *P. planifrons* abundance also appears to be influenced by substrate size, with a preference for cobbles rather than sand and boulders (Olsson et al. 2006). Similarly, Jowett et al. (2008) found *P. planifrons* abundance reduced as substrates became larger (e.g., dominated by boulders) and as streams became wider than 6 m.

In lakes, this species is generally found in the photic zone during the day but moves into the littoral zone to feed at night. Kōura can be found in deep waters (e.g., up to 180 m deep in Lake Taupō) and can be found in high elevation streams (e.g., on Mount Ruapehu at 1,260 m above sea level). Kusabs et al. (2015a) found that kōura abundance and distribution in seven Te Arawa lakes was influenced by the combined effects of lake-bed sediments, lake morphology, and hypolimnetic conditions related to trophic state. Sediment particle size was identified as the strongest driver of kōura abundance and biomass, with kōura populations increasing with increasing sediment particle size. Kōura abundance was highest in lakes Rotomā, Rotorua and Rotoiti which had a high proportion of coarse lake bed substrates and low in lakes Ōkāreka, Rotokākahi, Tarawera and Ōkaro where lake bed substrates were comprised mainly of mud (Kusabs et al. 2015a).

10.2 The fishery

The main agencies involved in the management of kōura are MPI (e.g., Fisheries Act 1996 and Biosecurity Act 1993) and DOC (e.g., Conservation Act 1987). There are no species-specific conservation measures in place for kōura. Various iwi around the country are progressing formal management arrangements to manage their kōura fisheries (e.g., Te Arawa). Currently, kōura may legally be gathered for personal consumption up to a limit of 50 crayfish per day. However, the selling, trading or possession of kōura for the purposes of sale or trade is currently illegal, with the exception of freshwater crayfish produced by aquaculture (see below). Any authorisations involving freshwater species (e.g., fish farming, transferring species) need to be approved by DOC, and in some cases agencies like MPI and iwi. For example, the **Ngāi Tahu Claims Settlement Act** prohibits the targeted commercial harvest of “Waikōura – freshwater crayfish (*Paranephrops spp.*)”. Te Roroa have a fisheries protocol with MPI that lists freshwater crayfish as a taonga species (MPI undated; Williams et al. 2017).

In the past, Māori actively managed the kōura fishery through a combination of approaches such as rāhui, ownership rights based on ancestral fishing grounds and selective harvesting. Occasional releases of kōura were made into waterways to boost populations (e.g., return of gravid females) ensured the long-term viability of the populations. As part of the **Te Arawa Lakes Settlement Act 2006** the Crown has made regulations to empower the Trustees of the Te Arawa Lakes Trust to manage the customary and recreational harvest of selected fisheries (including kōura) in fourteen Te Arawa Lakes, but not the streams and rivers flowing into the lakes. The Te Arawa Lakes (Fisheries) Regulations 2006 cover non-commercial customary fishing within the Te Arawa fisheries area and do not provide for commercial fishing. The Act provides for the establishment of Komiti Whakahaere to manage the customary fisheries in accordance with Te Arawa tikanga and kawa. The Komiti Whakahaere are in the process of developing the Mahire Whakahaere or Te Arawa Lakes Fisheries Plan which is required under the Regulations to provide for the sustainable management of customary fisheries in the Te Arawa lakes. Several customary management changes are suggested in Kusabs et al. (2015b) to protect and enhance the Te Arawa Lakes kōura fishery, including: (1) Restricting access to the fishery; (2) Implementation of a minimum legal length; (3) Implementing closed fishing seasons; and (4) Protecting egg-bearing and soft-shelled (moulting) kōura.

10.2.1 Aquaculture

Freshwater crayfish aquaculture is in an early development stage, with no farm currently producing large volumes of saleable stock (less than 500 kg combined total annual production in Aotearoa-New Zealand). The practice of harvesting of wild stocks for the seeding of aquaculture ventures, and the possibility of direct commercial harvest have fuelled concerns for the sustainability of targeted populations (Whitmore et al. 2000). In 2015 there were 17 licensed freshwater crayfish farms but only four (all in the South Island) were in production. All are selling on the domestic market. Market feedback indicates that there is export potential for kōura if consistent supply of large quantities can be achieved (Ernslaw One Ltd 2016). Ernslaw One’s initiative of farming kōura in the fire ponds of its South Island forests has recently received attention in the media (Tait-Jamieson 2017²¹).

²¹ <https://www.newsroom.co.nz/@living-room/2017/08/29/45024/sustainable-nz-crayfish-venture-wins-accolades>

10.3 Life cycle, age and growth

P. planifrons reproduce between April and December with peak reproduction occurring in May and June (Hopkins 1967). Females typically reach reproductive maturity between 17–20 mm orbit-carapace length (OCL) (Parkyn et al. 2002), fecundity increases with OCL²² (Kusabs et al. 2015a) and females can carry approximately 20–200 eggs each (Figure 10-2).



Figure 10-2: Stages in kōura life cycle. Source: Ian Kusabs, Karen Thompson.

The total duration of breeding from peak egg laying to the release of juveniles is estimated to be 28 weeks for the autumn-winter period, 19 to 20 weeks in spring-summer breeding groups for Northern lake populations (Devcich 1979), 25 to 26 weeks for Northern stream populations (Hopkins 1967), and up to 60 weeks for Southern kōura in stream populations (Whitmore 1997). Warmer water temperatures speed up the egg development process (Jones 1981a). A generic lifecycle for kōura populations living in lakes is presented in Figure 10-3.

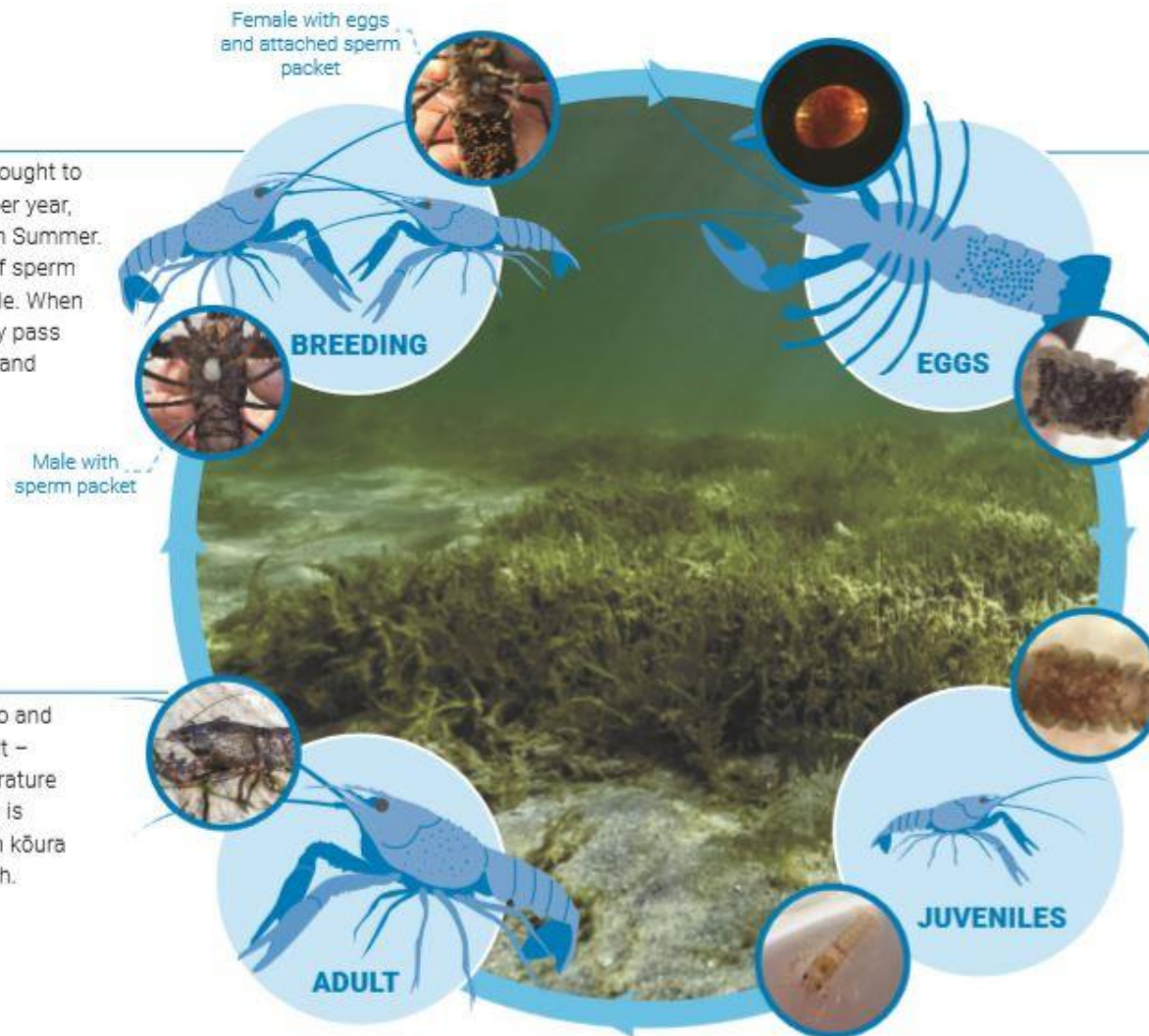
Incubation of eggs and young usually takes around 24 weeks but the duration is temperate dependent. In the Te Arawa Lakes, Kusabs et al. (2015a) found egg-bearing females throughout the year, although only occasionally during the summer months. Once the eggs hatch into juveniles, they are carried by the mother for up to three weeks (although this duration varies among females) and undergo two moults before they become independent (Hopkins 1967). The juveniles, typically at 4 mm carapace length, leave the mother in January/February.

Juveniles that enter the population in spring or early summer are likely to grow larger in their first year than those that leave the female in late summer as they have the advantage of growth through the summer months. *P. planifrons* is thought to mature in 18 months to 2 years in streams (Jones 1981b, Parkyn 2000), depending on temperature, while Devcich (1979) estimated that they probably matured in their third year in lakes.

²² Kōura length is determined using the Orbit-Carapace Length (OCL) which measured from behind the eye to the end of the carapace along the top and centre of the back.

Breeding

Kōura that live in lakes are thought to have two breeding seasons per year, one in late Autumn and one in Summer. Male kōura attach a packet of sperm on the underside of the female. When the female lays her eggs, they pass through the packet of sperm and become fertilised.



Adults

It can take kōura between two and four years to become an adult – seasonality and water temperature have an effect. *P. zealandicus* is slower growing. A fully-grown kōura averages 12 – 15 cm in length.

Eggs

A female can have anywhere between 20 – 200 eggs at a time, which are attached to hairs on the underside of the female's tail. The eggs can stay on the female anywhere between 4 and 15 months until they are fully developed into miniature kōura – the time it takes can vary depending on the water temperature and species.

Juveniles

Juvenile kōura cling to their mother's abdomen using their rear legs until they are large enough to defend themselves and live alone.

Figure 10-3: Kōura (*P. planifrons*) life cycle in lakes. Source: NIWA.

Kōura longevity is dependent on habitat conditions. Hopkins (1967) found *P. planifrons* did not generally survive beyond three years at study sites in the Wairarapa, while Parkyn et al. (2002) found *P. planifrons* in Waikato pasture streams reached four years old compared to seven years old in nearby native forest streams. Parkyn et al. (2002) found females reached the reproductive threshold of 17-20 mm OCL within one year in pasture streams in contrast to two years in forest streams where water temperatures were cooler. They observed *P. planifrons* (particularly juveniles) grew faster in pasture streams due to greater moult increments and frequency, as well as faster growth rates. Parkyn (2000) found few *P. planifrons* survived beyond their first year with high winter mortality rates especially in pasture (39%) compared to native forest (18%) streams. *P. planifrons* is considered a slow-growing, late maturing and very long-lived species (Furse et al. 2015). Key demographic estimates are in Table 10-1.

Table 10-1: Key demographic characteristics for kōura (*P. planifrons*). * denotes lake populations

Key demographic parameters	Estimates	References
Maximum size (OCL or RCL)	35 mm (OCL)	Parkyn and Kusabs (2007)
Minimum size sexual maturity (OCL or RCL ²³)	31 RCL (Female) 27 RCL (Male) 20 mm OCL	Parkyn et al. (2002)
Average size at maturity (mm)	–	–
Average age at maturity (yrs)	–	–
Longevity (yrs)	20*	Devcich (1979)
Growth parameter (K) ²⁴	–	–
Fecundity	20–30 eggs (17 mm carapace length) 150 eggs (30 mm carapace length)	–
Size _{leaving parent} (mm)	3.5	Hopkins (1967)
Mean egg size (mm)	–	–
Generation length (yrs)	Females reproduced every 2 years	–

10.4 Distribution and environmental preferences

P. planifrons are endemic to Aotearoa-New Zealand. They can be found at 500 metres above sea level and to a depth of more than 100 m (Furse et al. 2015). This species distribution extends from the far north of Aotearoa-New Zealand, south to Marlborough, Nelson, and the West Coast of the South Island (Figure 10-4). *P. planifrons* is found in several nearshore islands around Aotearoa-New Zealand, including Great Barrier, Great Mercury, Kapiti, and D'Urville Islands (McDowall 2005). Although kōura are fairly widespread and abundant in certain locations, there are also areas of the country where they are sparse and/or populations have established due to translocation (e.g., Lake Georgina in the upper Rakaia River catchment, McDowall 2005). Taranaki and Auckland regions appear to be population hotspots (Williams et al. 2017).

There are few observations of this species in the central North Island (with the exception of the Te Arawa and Tūwharetoa lakes), the East Cape, Canterbury and Fiordland regions. Kōura are also rare in low calcium waters, such as Lake Tikitapu. The absence of *P. planifrons* along the central to lower west coast of the South

²³ The RCL is the measurement from the rostrum to the carapace and is sometimes used instead of the OCL measurement

²⁴ Eel growth rates are linear (Graynoth and Taylor 2004).

Island is thought to be related to the enduring effects of glaciation during the Pleistocene on stream biota (McDowall 2005). The estimated extent of occurrence likely exceeds 45,200 km² (Parkyn et al. 2010).

P. planifrons can tolerate higher **temperatures** than the southern species (*P. zealandicus*), but optimum temperatures are likely to be less than 23°C. They can tolerate temperatures at 35°C but only for brief periods (Devcich 1979). In an experiment to simulate exposure to high temperatures in pasture streams, 100% of crayfish exposed to 26.2°C survived after 24 hours compared to 5-10% at 28.9°C and 0% at 32.4°C. When hypoxia stress was added to five days of exposure to 26.2°C (to simulate deoxygenation in eutrophic lowland streams and as a multiple stressor) survival decreased to 40%, compared to 95% in crayfish held at 19°C (Albert et al. 2015).

P. planifrons is considered a cool water species, and in the central North Island inhabits areas where water temperatures fluctuates seasonally from ~6–18°C in native forest habitat and ~5–25° C in pastoral habitats (Parkyn et al. 2002; Furse et al. 2015). Temperature is considered the primary determinant of growth in *P. planifrons* (Parkyn et al. 2002) and also influences their mobility. Instream movements are greater in the summer and in pasture streams compared with colder forest streams (Parkyn 2000). Furse et al. (2015) suggest a dedicated evaluation of the thermal limit of this species might be prudent given predicted increases in environmental temperatures associated with climate change.

Kōura in lakes can be affected by periods of anoxia, e.g., they are now absent from Lake Ōkaro as this lake has no oxygen in its bottom waters during summer. During studies of acute lethality of low **dissolved oxygen**, *P. planifrons* held for 48 hours at 15°C showed a LC50 of 0.77 ± 0.06 mg/L, suggesting tolerance for low dissolved oxygen conditions (Landman et al. 2005). Animals in the study were denied access to the water surface and were unable to leave the tanks so tolerance for low oxygen conditions may be lower if access to such additional oxygen sources had been available.

Habitat preferences are for still or slow **flowing water** and depths of 0.2-0.3 m s⁻¹ in streams (Jowett et al. 2008). Koura can survive periods not being submerged in water.

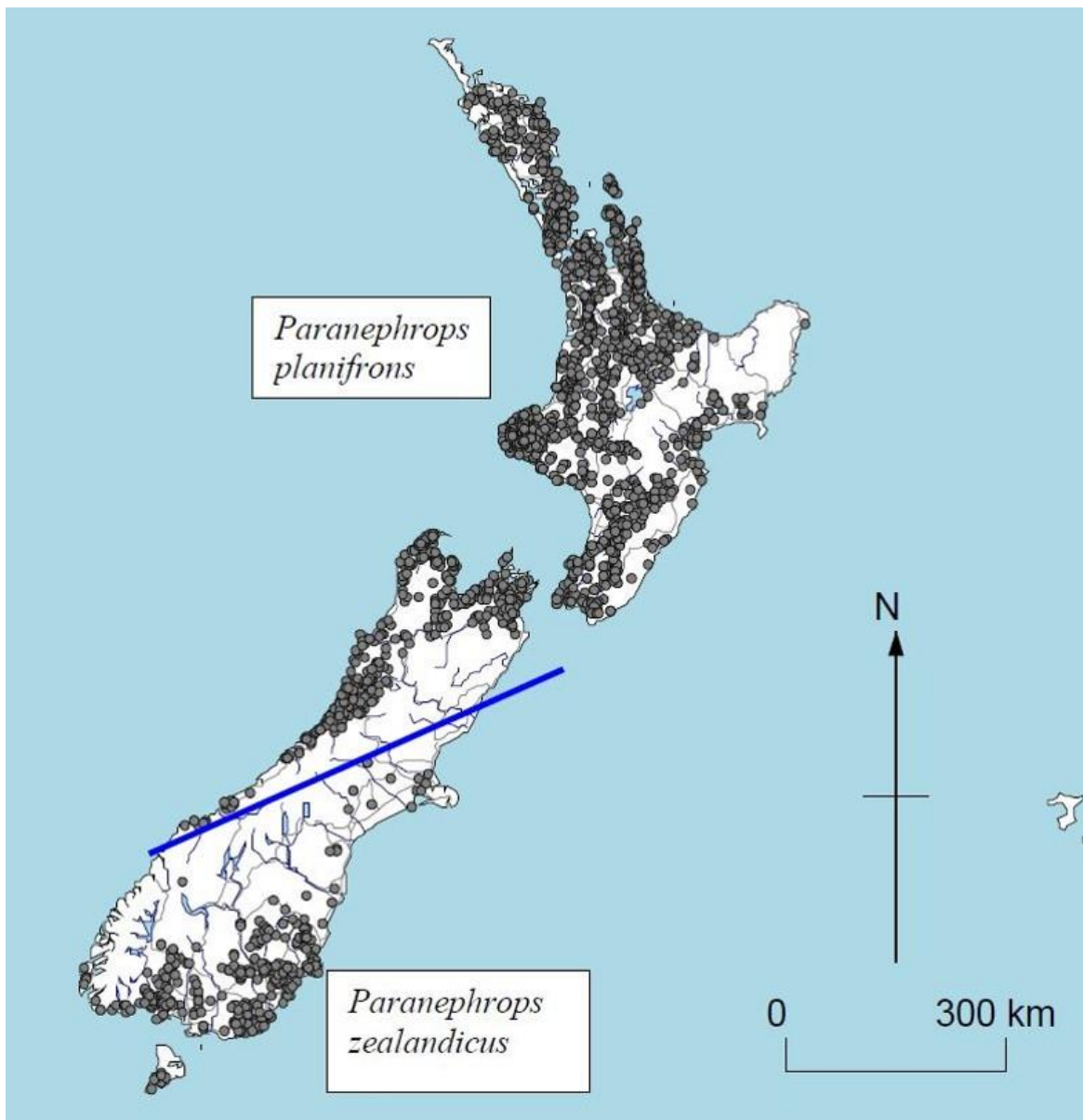


Figure 10-4: Distribution of Aotearoa-New Zealand's endemic kōura species *Paranephrops planifrons* (Northern species) and *Paranephrops zealandicus* (Southern species) are separated by the Southern Alps. Source: Parkyn and Kusabs (2007).

10.5 Diet and predation

P. planifrons are omnivorous, feeding predominantly on benthic invertebrates and leaf detritus (Parkyn et al. 2001). Snails, chironomids and mayfly larvae are important components of their diet (Whitmore et al. 2000; Hollows et al. 2002). Feeding in lakes tends to be concentrated in the littoral zone where more food is often found.

There are ontogenetic shifts in diet. Juveniles mainly consume invertebrates while adults fed mainly on detritus (Parkyn et al. 2002). Parkyn et al. (2002) found the proportion of each food type consumed by *P. planifrons* differed between land-use which was linked to a variety of factors including habitat choice, available food sources and also likelihood of predator avoidance behaviour (Kelly 2019).

Kōura make up a large proportion of catfish diet in Lake Taupō (up to 80% in some areas). Anecdotally, the introduction of perch to a Northland lake has dramatically decreased crayfish populations. In some South

Island streams, brown trout predate on koura between 7–20 mm carapace length²⁵ and make up 6% of their diet (Blair et al. 2012). Other known predators are rainbow trout (*Oncorhynchus mykiss*; Blair et al. 2012) and eels (*Anguilla australis* and *Anguilla dieffenbachii*; Hicks and McCaughan 1997). Freshwater crayfish are important to diet of birds such as inland cormorants (Dickinson 1951).

10.6 Recruitment

P. planifrons have weak dispersal abilities and there is little inter-catchment exchange, except where there are downstream freshwater connections. Translocation of populations is required to restore areas and should preferably be from within catchments (Smith and Smith 2009). Once crayfish have been made locally extirpated, it is difficult for recolonisation to occur naturally (S. Parkyn, pers. comm. 2008).

10.7 Current impacts of climate change

A recent review of freshwater crayfish found that globally, 87% of crayfish species are highly sensitive to climate change (primarily due to habitat specialisation), 35% have low adaptive capacity and 57% are highly exposed (Hossain et al. 2018). Hossain et al. (2019) included *P. planifrons* in their global Climate Change Vulnerability Assessment dataset. They found that *P. planifrons* was vulnerable to climate change based on relative concentration pathway scenarios of 6.0 and 8.5 by the year 2070. However, the environmental variable they are exposed to (i.e., wind, air temperature) was not specified.

P. planifrons may be sensitive to flood spates (Parkyn and Collier 2004), and any increase in severe weather events due to changes in climatic and weather patterns (Furse et al. 2015). Flood spates are considered an emerging threat to this species (Furse et al. 2015).

10.8 Ecosystem level interactions

The interactions (with competitors, prey/predators, host/parasites, mutualists etc) between freshwater crayfish the wider ecosystem include:

- Freshwater crayfish have the potential to play significant roles in energy transfer in stream and lake food webs through their varied role as detritivores, herbivores and carnivores (Momot 1995), and through their influence on biomass and species composition of invertebrates and macrophytes (Nystrom et al. 1996);
- *P. planifrons* can act as a keystone species where they occur in high densities through influences on invertebrate community structure directly via predation, or indirectly by influencing organic matter processing rates, and bioturbating benthic sediment (Parkyn et al. 1997; Parkyn et al. 2001);
- Parkyn et al. (2001) found that *P. planifrons* represented a top down predator on invertebrates but were below eels in the trophic hierarchy. Parkyn et al. (1997) suggests that loss of *P. planifrons* from streams could result in an invertebrate community structure alteration, particularly in pool habitats where deposition of leaf matter and fine sediment accumulation are highest;
- *P. planifrons* are shredders and can contribute to both particulate organic matter production and leaf processing in streams. They can also have indirect effects on stream community structure through bioturbation and excretion (Furse et al. 2015).

10.9 Additional (multiple) stressors

The stressors derived from the IUCN threat assessment (n=5), included 'dams and water management/use [surface water abstraction]'; 'Invasive and other problematic species/genes [problematic native species,

²⁵ https://niwa.co.nz/our-science/freshwater/tools/kaitiaki_tools/species/koura

introduced genetic material]; pollution [domestic and urban wastewater]’ and ‘transportation and service corridors [shipping lanes]’ (see Supplementary Appendix 2).

10.10 Critical data gaps and level of uncertainty

An important component of a CCVA is the data quality and specificity of information on which it is based. Data quality for each sensitivity attribute was characterised based on the type of data/information used. Assessments of data quality took the following into account:

- *Paranephrops planifrons* has been assessed as ‘Least Concern’ (Grainger et al. 2018). However, this species is impacted upon locally by habitat degraded through bank erosion, stream channel works and water pollution. There is also a gradual decline in numbers potentially as a result of land use change introduced species and subsistence level harvesting for human consumption. Further research is needed to determine the abundance of this species, and whether it is being impacted upon by any major threat processes on a global scale (Parkyn 2010);
- The taxonomy of *Paranephrops* is highly uncertain. The recognition and conservation of genetic diversity is important, as it allows populations and species to persist through changing environments over evolutionary timescales (Jamieson et al. 2008; Smith 2009) (i.e., it enables differing responses to ongoing and future global change, disease outbreaks, etc.);
- *Paranephrops* taxonomy is more complex than the long-accepted scenario of two distinct species (Apte et al. 2007). Previously, it was believed that two species of kōura/kēwai existed in Aotearoa-New Zealand, which are separated by the Southern Alps: *P. planifrons* in the North Island and in the northwest of the South Island, and *P. zealandicus* along the eastern side of the South Island and on Stewart Island. Apte et al. (2007), using a mitochondrial DNA marker, noted a third major lineage of kōura were present on the South Island (mid to northern West Coast), and that this cryptic (genetically distinct but morphologically similar species) West Coast group appeared to be more closely related to *P. zealandicus* than to *P. planifrons*;
- A subsequent study by Smith and Smith (2009) in the central-west North Island also recorded high levels of genetic diversity in *P. planifrons* over small spatial scales among neighbouring catchments. This study, also using mitochondrial DNA markers (COI), found that *P. planifrons* had a greater proportion of the total genetic diversity distributed among catchments (c. 72%) and markedly less diversity within populations (c. 18%). Streams separated by less than 10 km, but in different catchments, were characterised by catchment-specific haplotypes – Apte et al. (2007) similarly recorded a large diversity of haplotypes. Smith and Smith (2009) also noted that it was unlikely there had been significant translocations among central west North Island catchments, as they found catchment-specific haplotypes in seven out of the nine catchments studied. These studies, especially the latter, indicate that natural between-catchment movement of kōura is limited due to their low dispersal capability.

10.11 References

- Albert, A.M., Clearwater, S.J., Hickey, C.W. (2015) Multiple stressors: tolerance of juvenile crayfish *Paranephrops planifrons* to acutely elevated water temperatures, followed by hypoxic stress. *SETAC Australasia Meeting*, Nelson, New Zealand. 26-28 August 2015.
- Apte, S., Smith, P.J., Wallis, G.P. (2007) Mitochondrial phylogeography of New Zealand freshwater crayfishes, *Paranephrops* spp. *Molecular Ecology*, 16: 1897–1908.
- Blair, J.M., Hicks, B.J., Pitkethley, R., Ling, N. (2012) Diet of rainbow trout in Lake Rotoiti: an energetic perspective. *New Zealand Journal of Marine and Freshwater Research*, 46: 557–565.

- Chapman, M.A., Lewis, M.H., Winterbourn, M.J. (2011) *Guide to the freshwater crustacea of New Zealand*. Christchurch, New Zealand Freshwater Sciences Society.
- Devcich, A.A. (1979) An ecological study of *Paranephrops planifrons* (White) (Decapoda: Parastacidae) in Lake Rotoiti, North Island, New Zealand. Unpublished PhD thesis, University of Waikato, Hamilton, New Zealand.
- Dickinson, P. (1951) Stomach contents of New Zealand inland shags. *Australian Journal of Marine and Freshwater Research*, 1: 245–253.
- Ernslaw One Ltd. (2016) Freshwater Crayfish Farming. A Guide to Getting Started. 28 p. Available online via: <http://keewai.co.nz/wp-content/uploads/2016/07/Kōura-Guide-Final-Version.pdf>
- Furse, J., Burnham, Q.F., Dawkins, K., Richardson, A.M.M. (2015) Freshwater Crayfish: A Global Overview, edited by Tadashi Kawai, et al., CRC Press LLC, 2015. ProQuest Ebook Central. <http://ebookcentral.proquest.com/lib/miami/detail.action?docID=2122555>.
- Grainger, N., Harding, J., Drinan, T., Collier, K., Smith, B., Death, R., Makan, T., Rolfe, J. (2018) Conservation status of New Zealand freshwater invertebrates, 2018. *New Zealand Threat Classification Series 28*. Wellington: Department of Conservation: 25.
- Hicks, B.J., McCaughan, H.C. (1997) Land use, associated eel production, and abundance of fish and crayfish in streams in Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 31:635–650.
- Hollows, J.W., Townsend, C.R., Collier, K.J. (2002) Diet of the crayfish *Paranephrops zealandicus* in bush and pasture streams: insights from stable isotopes and stomach analysis. *New Zealand Journal of Marine and Freshwater Research*, 36:129–142.
- Hopkins, C.L. (1967) Breeding in the freshwater crayfish *Paranephrops planifrons* White. *New Zealand Journal of Marine and Freshwater Research*, 1:51–58.
- Hossain, M.A., Lahoz-Monfort, J.J., Burgman, M.A., Böhm, M., Kujala, H., Land, L.M. (2018) Assessing the vulnerability of freshwater crayfish to climate change. *Diversity and Distributions*, 24: 1830–1843.
- Hossain, M.A., Kujala, H., Bland, L.M., Burgman, M., Lahoz-Monfort, J.J. (2019) Assessing the impacts of uncertainty in climate-change vulnerability assessments. *Diversity and Distributions*, 25: 1234–1245.
- Jamieson, I.G., Grueber, C.E., Waters, J.M., Gleeson, D.M. (2008) Managing genetic diversity in threatened populations: a New Zealand perspective. *New Zealand Journal of Ecology*, 32: 130–137.
- Jones, J.B. (1981a) Growth of two species of freshwater crayfish (*Paranephrops* spp.) in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 15: 15–20.
- Jones, J.B. (1981b) The aquaculture potential of New Zealand freshwater crayfish. *New Zealand Agricultural Science*, 15: 21–23.
- Jowett, I., Parkyn, S., Richardson, J. (2008) Habitat characteristics of crayfish (*Paranephrops planifrons*) in New Zealand streams using generalised additive models (GAMs). *Hydrobiologia*, 596: 353–365.

- Kelly, J. (2019). Comparison of kōura (*Paranephrops planifrons*) populations in Waikato hill-country streams with varying type and age of riparian planting. MSc thesis, University of Waikato, Hamilton, New Zealand. Retrieved from <https://hdl.handle.net/10289/12934>
- Kusabs, I.A., Quinn, J.M., Hamilton, D.P. (2015a) Effects of benthic substrate, nutrient enrichment and predatory fish on freshwater crayfish (kōura, *Paranephrops planifrons*) population characteristics in seven Te Arawa (Rotorua) lakes, North Island, New Zealand. *Marine and Freshwater Research*, 66: 631–643.
- Kusabs, I.A., Hicks, B.J., Quinn, J.M., Hamilton, D.P. (2015b) Sustainable management of freshwater crayfish (kōura, *Paranephrops planifrons*) in Te Arawa (Rotorua) lakes, North Island, New Zealand. *Fisheries Research*, 168: 35–46.
- Landman, M.J, Van Den Heuvel, M.R., Ling, N. (2005) Relative sensitivities of common freshwater fish and invertebrates to acute hypoxia. *New Zealand Journal of Marine and Freshwater Research*, 39: 1061–1067. DOI: 10.1080/00288330.2005.9517375
- McDowall, R.M. (2005) Historical biogeography of the New Zealand freshwater crayfishes (Parastacidae, *Paranephrops* spp.): Restoration of a refugial survivor? *New Zealand Journal of Zoology*, 32: 55–77.
- Ministry for Primary Industries (MPI). (undated). Fisheries Plan Freshwater Species. 28 p. Available online via: <http://fs.fish.govt.nz/Doc/16403/FreshwaterSpeciesDraft%20FishPlan%20-%20Current%20State.pdf.ashx>
- Momot, W.T. (1995) Redefining the role of crayfish in aquatic systems. *Reviews in Fisheries Science*, 3: 33–63.
- Nyström, P., Brönmark, C., Granéli, W. (1996) Patterns in benthic food webs: a role for omnivorous crayfish. *Freshwater Biology*, 36: 631–646.
- Olsson, K. (2003) The influence of introduced brown trout (*Salmo trutta*) on the distribution and abundance of native crayfish (*Paranephrops planifrons*) and galaxiids (*Galaxias* spp.) in New Zealand streams. Honours thesis in Environmental science, Department of Ecology, Limnology, Lund University, Sweden.
- Olsson, K., Stenroth, P., Nyström, P., Holmqvist, N., McIntosh, A.R., Winterbourn, M.J. (2006) Does natural acidity mediate interactions between introduced brown trout, native fish, crayfish and other invertebrates in West Coast New Zealand streams? *Biological Conservation*, 130: 255–267.
- Parata, R.N. (2019) Quantifying relationships between kōura (*Paranephrops planifrons*) abundance and micro-habitat features in Waikato hill-country streams. MSc thesis, University of Waikato, Hamilton, New Zealand. Retrieved from <https://hdl.handle.net/10289/13086>
- Parkyn, S.M. (2000) Effects of native forest and pastoral land use on the population dynamics and trophic role of the New Zealand freshwater crayfish *Paranephrops planifrons* (Parastacidae). Unpublished PhD thesis, University of Waikato, New Zealand.
- Parkyn, S. (2010) *Paranephrops planifrons*. The IUCN Red List of Threatened Species 2010: e.T153750A4540571. <http://dx.doi.org/10.2305/IUCN.UK.2010-3.RLTS.T153750A4540571.en>
- Parkyn, S.M., Collier, K.J. (2004) Interaction of press and pulse disturbance on crayfish populations: flood impacts in pasture and forest streams. *Hydrobiologia*, 527:113–124.
- Parkyn, S., Kusabs, I. (2007) Taonga and mahinga kai species of the Te Arawa lakes: a review of current knowledge – kōura. *NIWA Client Report HAM2007-022*. NIWA, Hamilton: 35.

- Parkyn, S.M., Rabeni, C.F., Collier, K.J. (1997) Effects of crayfish (*Paranephrops planifrons*: Parastacidae) on in-stream processes and benthic faunas: A density manipulation experiment. *New Zealand Journal of Marine and Freshwater Research*, 31:685–692.
- Parkyn, S.M., Collier, K.J., Hicks, B.J. (2001) New Zealand stream crayfish: functional omnivores but trophic predators? *Freshwater Biology*, 46: 641–652. doi:10.1046/j.1365-2427.2001.00702.x
- Parkyn, S.M., Collier, K.J., Hicks, B.J. (2002) Growth and population dynamics of crayfish *Paranephrops planifrons* in streams within native forest and pastoral land uses. *New Zealand Journal of Marine and Freshwater Research*, 36: 847–861.
- Smith, P.J. (2009) Genetic principles for freshwater restoration in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 43: 749–762.
- Smith, P.J., Smith, B.J. (2009) Small-scale population-genetic differentiation in the New Zealand caddisfly *Orthopsyche fimbriata* and the crayfish *Paranephrops planifrons*. *New Zealand Journal of Marine and Freshwater Research*, 43: 723–734.
- Tait-Jamieson, A. (2017) Sustainable crayfish venture thrives in NZ forests. *NZ Life & Leisure*. 30 August 2017.
- Whitmore, N. (1997) Population ecology of the freshwater crayfish *Paranephrops zealandicus* and its effect on the community structure of a lowland bush stream. Unpublished MSc thesis, University of Otago, Dunedin, New Zealand.
- Whitmore, N., Huryn, A.D., Arbuckle, C.J., Jansma, F. (2000) Ecology and distribution of the freshwater crayfish *Paranephrops zealandicus* in Otago. *Science for Conservation*, 148: 42.
- Williams, E., Crow, S., Murchie, A., Tipa, G., Egan, E., Kitson J., Clearwater, S., Fenwick, M. (2017) Understanding Taonga Freshwater Fish Populations in Aotearoa-New Zealand. *NIWA Client Report No. 2017326HN*. NIWA Wellington.

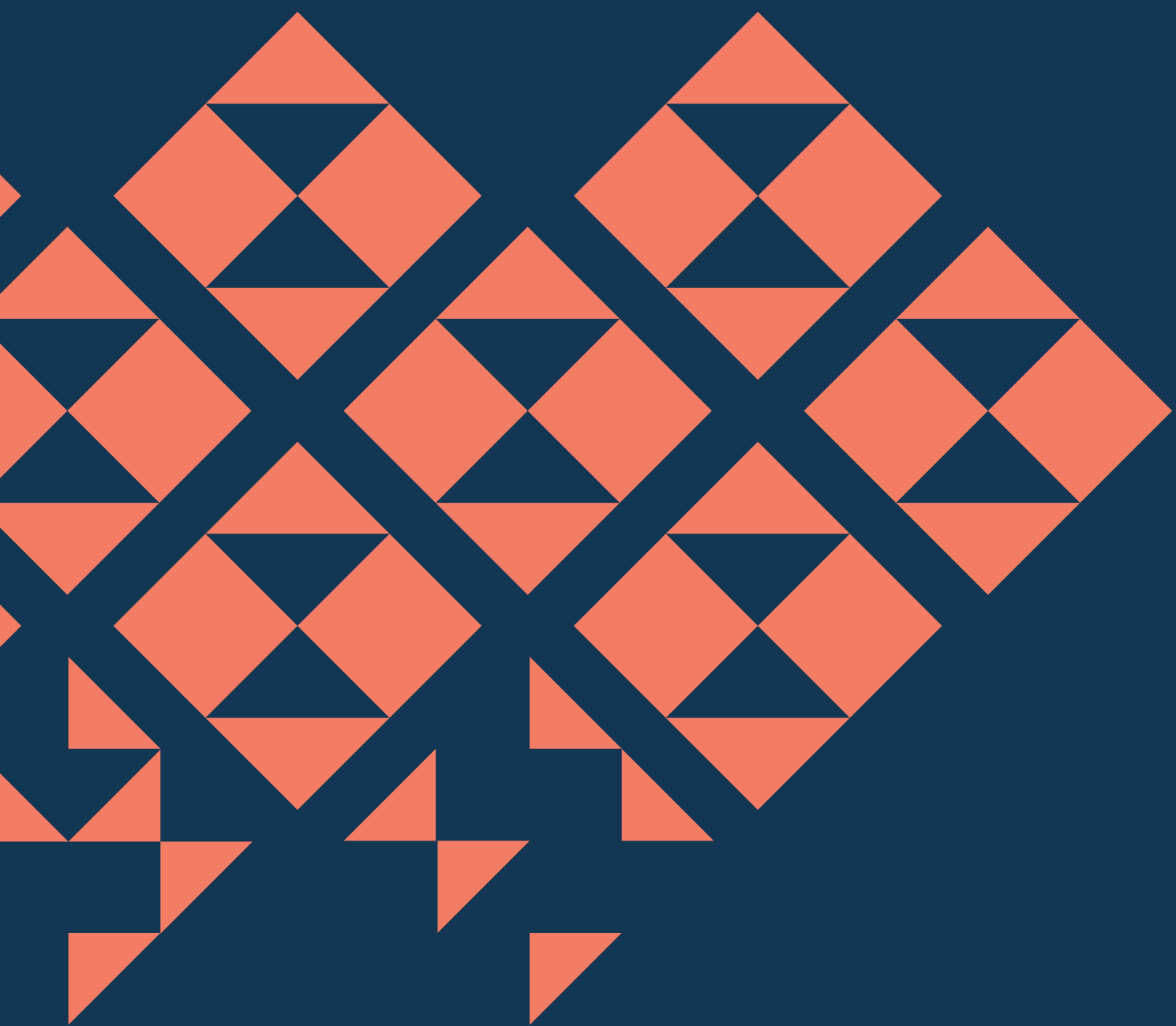
Appendix A Environmental variables associated with taonga fish species distribution

Table A-1: Environmental variables associate with taonga fish species distributions. Source: Adapted by Egan from Leathwick et al. (2008). Note: yellow-eye mullet, kākahi [*E. menziesii*] and kōura [*P. planifrons*] were not included in the Leathwick et al. (2008) distribution models. Blue shading denotes environmental variables that explained >5% of the variation in species distribution.

Predictor	<i>Anguilla australis</i>	<i>Anguilla dieffenbachii</i>	<i>Galaxias argenteus</i>	<i>Galaxias brevipinnis</i>	<i>Galaxias fasciatus</i>	<i>Galaxias maculatus</i>	<i>Geotria australis</i>
<i>DSAvgSlope</i>	9.5	5.0	4.6	9.1	3.7	12.7	31.5
<i>DSDist</i>	3.7	3.8	5.4	4.1	26.6	23.7	8.1
<i>SegSumT</i>	22.8	13.4	4.8	4.8	5.8	5.5	5.3
<i>DSMaxSlope</i>	13.9	8.0	5.9	3.2	2.2	12.1	5.1
<i>USRainDays</i>	2.2	5.5	11.7	14.0	4.6	2.7	4.1
<i>Method</i>	7.1	11.2	9.6	1.9	3.5	4.3	2.0
<i>ReachSubstrate</i>	3.0	6.4	3.0	4.7	2.4	3.6	4.5
<i>SegTSeas</i>	2.2	6.7	4.1	4.1	5.4	3.7	3.8
<i>SegFlowStability</i>	2.3	2.8	7.0	3.1	2.6	2.9	3.1
<i>SegLowFlow</i>	1.6	1.9	2.7	2.6	5.2	3.8	3.3
<i>SegShade</i>	2.4	1.9	3.1	3.3	11.1	1.7	2.3
<i>SegN</i>	5.6	2.2	2.9	5.6	1.8	2.5	2.4
<i>USSlope</i>	3.0	3.0	2.7	6.9	2.6	2.1	3.3
<i>USAvgT</i>	1.3	1.8	4.6	2.5	1.9	2.5	2.8

Table A-2: Continued.

Predictor	<i>Anguilla australis</i>	<i>Anguilla dieffenbachii</i>	<i>Galaxias argenteus</i>	<i>Galaxias brevipinnis</i>	<i>Galaxias fasciatus</i>	<i>Galaxias maculatus</i>	<i>Geotria australis</i>
<i>SegSlope</i>	2.3	1.9	5.6	5.8	4.6	2.6	2.7
<i>USPhosphorus</i>	2.5	2.2	2.9	2.7	3.0	2.2	2.4
<i>ReachHabitat</i>	2.2	2.0	4.4	3.4	2.7	2.0	2.2
<i>USHardness</i>	1.7	2.7	3.0	2.2	1.5	1.8	3.4
<i>USNative</i>	4.4	1.5	3.0	4.3	2.1	1.5	2.1
<i>USCalcium</i>	2.1	1.1	1.9	1.8	3.2	1.1	1.8
<i>Tcos</i>	1.9	2.9	1.4	1.2	0.8	3.0	1.7
<i>Tsin</i>	1.3	0.6	2.0	1.0	0.7	1.2	1.6
<i>DSDist2Lake</i>	0.3	0.9	2.3	7.1	1.5	0.1	0.2
<i>DSDam</i>	0.1	9.9	0.0	0.2	0.0	0.0	0.0
<i>USPeat</i>	0.2	0.3	1.6	0.1	0.3	0.7	0.1
<i>USLake</i>	0.4	0.5	0.1	0.2	0.1	0.2	0.1
<i>USGlacier</i>	0.0	0.0	0.0	0.3	0.0	0.0	0.0



NIWA
Taihoro Nukurangi



Te Wai Māori