PFAS contamination in freshwater food sources: A study on Aotearoa New Zealand's freshwater fish

A thesis submitted in partial fulfilment of

the requirement for the degree of

Master of Science in Ecology

at Massey University, Palmerston North, New Zealand

By

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2026

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Acknowledgements

From the beginning this research has been a labour of love. As with all meaningful endeavours, it has not been without challenges, and I am sincerely grateful to all who supported me along the way.

My sincere thanks go to my supervisors, Professor Alastair Robertson and Associate Professor Maria Minor who despite perhaps thinking I was crazy at times, trusted me enough to let me carry on regardless.

I am also grateful to Emeritus Professor Russell Death to whom I offer my heartfelt thanks for asking the hard questions, for his unwavering support in the face of the many challenges that came with this research, and for having the good sense to never tell me something wasn't possible.

I will be forever grateful to Associate Professor Brad Clarke, Jordan Partington, and the team at the University of Melbourne Australian Laboratory for Emerging Contaminants for the opportunity to learn and grown under their tutorship. My time in the ALEC lab was formative, professionally and personally.

This research was made possible through a Master's Scholarship funded by the Estate of Dr Eileen Fair. Dr Fair, who passed away early 2024, was a glass ceiling breaker whose footsteps I am honoured to follow in.

I am thankful to Dr Matthew Silver for inspiring my initial interest into PFAS world and for your invaluable support in the early stages of my research.

Thank you to the team at Traverse Environmental for keeping me employed whilst also giving me the flexibility to pursue my studies. I'd particularly like to thank Fiona Death for being my sounding board and voice of reason.

I extend my gratitude to the anglers from Manawatū Anglers club, Auckland Fly Fishing Club, Wellington Fly Fishing Club, and Kapitī Fly Fishing Club for providing fish liver samples, and

for their enthusiasm. Jim and Steve deserve particular thanks for their energy, excellent coffee, and life advice.

A heartfelt thank you to the rūnanga who supported this kaupapa: Te Atiawa ki Kapitī, Ngāti Kahangungu ki Wairarapa, and Ngāti Apa. I am grateful for your willingness to share your knowledge and for your assistance with sampling. Special thanks to Sam Ludden, whose enthusiasm is infectious.

To the McIntosh-Pearce family, thank you for listening, guiding and supporting me.

Thank you to my family, who understood little of this work but gave everything to support it, and me. In particular my brother, who undoubtedly had had better things to do with his weekends than setting and pulling in eels nets but chose to help me anyway.

Finally, to Theo, eel dissection assistant, snack provider, and calm amid my constant chaos thank you for being anything I needed.

Abstract

PFAS (per and polyfluoroalkyl substances) have become major potential toxicants of humans and wildlife over the last decade. PFAS are a class of environmentally persistent synthetic chemicals that have been detected in groundwater, drinking water supplies, freshwater fish, and even marine mammals. This has raised concern in many communities that rely on these freshwater species as food sources. Although PFAS concentrations have been assessed in some New Zealand waterways and marine mammals, surprisingly, only one study, limited to two eel samples has examined PFAS levels in commonly consumed New Zealand freshwater fish.

To investigate distribution, concentration and composition of PFAS across different locations and fish species, I collected six fish species across seven sites North Island, New Zealand. Livers and fins from wild-caught fish provided by local Salmonid anglers, tangata whenua (indigenous communities), and myself were analysed for 75 different PFAS.

Native shortfin eels *Anguilla australis* (Richardson, 1841), a culturally significant food source for Māori, were found to have higher total PFAS. In some samples, levels in the liver exceeded Food Safety Australia and New Zealand (FSANZ) trigger values. Concentrations of perfluorooctanesulfonic acid (PFOS) in the fish fins were strongly correlated with those in the fish liver, indicating that fins may serve as a useful non-lethal proxy for assessing PFOS contamination in fish. Overall, the results highlight the importance of addressing PFAS contamination in New Zealand's freshwater environments at a wider scale and signal potential consequences for public health, long-term sustainability of fisheries, and viability of export markets in New Zealand.

Myself and staff from Te Atiawa rūnanga pulling in fyke nets at Site E.
Photo Credit: Russell Death.

1 Chapter 1: Introduction

Freshwater fish species are an important, often culturally significant, food resource for many indigenous peoples and recreational fishermen worldwide (Noble *et al.*, 2016; Junqué *et al.*, 2025). Environmental contamination impacts fish health and growth, decreasing fecundity and ultimately leading to declines in population size (Wannas & Mohammed, 2025). The increasing presence of per- and polyfluoroalkyl substances (PFAS) in surface waters can adversely affect fish residing in contaminated waterways and poses potential risks for fish consumers, especially in regions where aquatic ecosystems are important food sources for local communities (Christensen *et al.*, 2022; Wei *et al.*, 2024). PFAS are a class of synthetic chemicals, characterized by their extreme environmental persistence, often referred to as "forever chemicals" (Giesy & Kannan, 2001). There is a growing body of evidence linking these compounds to adverse human and ecosystem health effects (Schrenk *et al.*, 2020; Fenton *et al.*, 2021; Panieri *et al.*, 2022). PFAS also exhibit biomagnification within food webs, resulting in elevated exposure risk for apex predators, including humans (Teunen *et al.*, 2021; Munoz *et al.*, 2022). Dietary intake of fish and seafood is a major pathway for human PFAS exposure (Hu *et al.*, 2018), and human populations that rely heavily on fish and seafood as dietary staples, such as many indigenous communities, may therefore experience disproportionately greater exposure compared to the general population (Cisneros-Montemayor *et al.*, 2016). Recreational fishing holds significant commercial, cultural, and social values for the island nation of Aotearoa New Zealand but the impact that PFAS has on these values is unknown, despite PFAS having been detected in New Zealand's waters and marine mammals (Stockin *et al.*, 2021; Close *et al.*, 2023).

PFAS Chemical Structure and Use

The persistence of PFAS in the environment can largely be attributed to their chemical structure (Evich *et al.*, 2022). The OECD (Organisation for Economic Co-operation and

Development) characterises PFAS as substances with at least a fully fluorinated methyl or methylene carbon atom (Organisation for Economic Co-operation Development (OECD), 2021). In most PFAS these atoms are arranged in carbon-fluorine chains, which vary in length depending on the specific PFAS compound; long chain compounds (C > 6) are generally regarded as more persistent in the environment than their short chain (C < 6) counterparts (Li *et al.*, 2020). These strong carbon fluorine bonds are among the strongest in organic chemistry and prevent PFAS from degrading in the environment (Itumoh et al., 2024). The OECD classification encompasses more than 1400 compounds with current and/or historic industrial and commercial uses, including in metal plating, stain and water protection, and in aqueous film-forming foams (AFFFs) (Place & Field, 2012). Poor management practices surrounding the use or production of these chemicals, have led to high volumes of PFAS being released into the environment at sites where they were manufactured, used and/or disposed of (Sznajder-Katarzyńska *et al.*, 2019). Globally, PFAS contamination is frequently detected near landfills used for chemical disposal, near airports, military bases, and fire stations that have frequently discharged AFFF's, and near chemical manufacturing plants (Ehsan *et al.*, 2024).

Despite concerns about their environmental and human health impacts, toxicity data and knowledge on environmental occurrence is limited to a small selection of Perfluoroalkyl acids (PFAA's) (Evich *et al.*, 2022). These include legacy compounds, Perfluorooctanoic acid (PFOA) and Perfluorooctanesulfonic acid (PFOS). Driven by concerns about their impact on human and environmental health, regulations governing the use and production of PFOA and PFOS, although varying in their stringency across different countries, are becoming increasingly prevalent. However, as a class, PFAS remain poorly regulated which has led to the use and development of novel PFAS as replacements for legacy compounds such as polyfluoroalkyl phosphate esters (PAP's) with equally unknown environmental and health impacts (Cousins *et al.*, 2020; De Silva *et al.*, 2021).

Consequences of PFAS for Aquatic Ecosystems

Aquatic ecosystems are often the endpoint for PFAS contamination. PFAS are highly mobile in water and can migrate into surface water bodies through stormwater run-off or seep into ground water though the soil leading to an accumulation in those waterbodies (Dauchy *et al.*, 2019; Christensen *et al.*, 2022). To fully comprehend the ecological consequences of PFAS, it's essential to understand how these compounds bioaccumulate and biomagnify within ecosystems (Ankley *et al.*, 2021; Brown *et al.*, 2023).

Bioaccumulation refers to the process by which organisms absorb and accumulate substances, such as PFAS, from their environment over time (**Fig. 1**). This accumulation occurs because the rate of intake exceeds the rate of excretion or metabolism (Sun *et al.*, 2022). **Biomagnification** describes the increasing concentration of these substances as they move up the food chain (Munoz *et al.*, 2022) (**Fig. 1**). Ecological factors, such as an organisms feeding guild or habitat preference, and environmental factors such as substrate type or water temperature, play a significant role in the bioaccumulation and biomagnification of PFAS in aquatic organisms (Arulananthan *et al.*, 2025; Roy *et al.*, 2025). Top predators have been found to have higher concentrations of PFOS than primary consumers (Chen et al., 2023; Giari et al., 2023). Benthic dwelling organisms often have higher concentrations of PFAS, particularly PFOS, as it binds to and accumulates in sediments (Sands et al., 2024). However, whilst numerous studies examine the factors influencing PFAS levels, results are often contradictory and/or only refer to one group of PFAS or a singular compound. (Miranda *et al.*, 2022; Nayak *et al.*, 2023).

Bioaccumulation within animals has been documented in a large number of PFAS including the legacy compounds PFOS and PFOA (Teunen *et al.*, 2021; Munoz *et al.*, 2022). The bioaccumulation rate varies among differing PFAS and is strongly influenced by carbon chain length, the PFAS functional head group, and species-specific ecology (Babut *et al.*, 2017). In general, long-chain PFAS exhibit higher bioaccumulation than their shorter chain counterparts, resulting in more frequent detection in animal tissues (Xie *et al.*, 2024). In

addition to chain length, PFAS stratification in the water column combined with species-specific ecology, can contribute to differences in the concentration and composition of PFAS within food webs. Their amphipathic nature, characterised by hydrophilic head groups and hydrophobic tail groups, results in PFAS accumulation at the air-water interface and within sediments, resulting in comparatively lower concentrations in the water column (Brusseau & Glubt, 2021) (**Fig. 2.**). Additionally, longer-chain compounds more readily accumulate in sediment compared to shorter-chain compounds, which tend to be more concentrated in surface water (Goodrow *et al.*, 2020). Such distribution patterns shape exposure pathways within aquatic food webs.

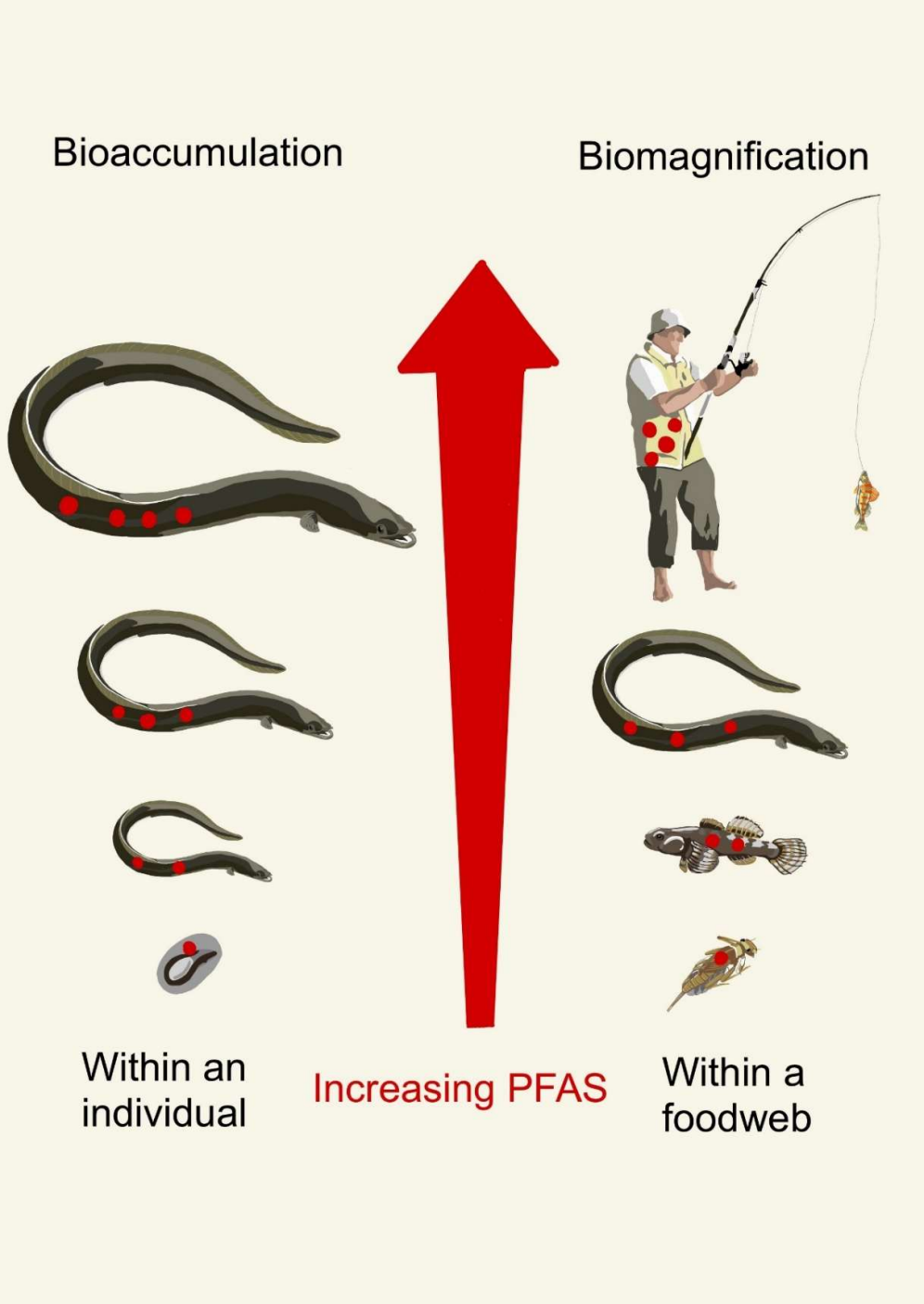

Figure 1: Conceptual illustration of bioaccumulation and biomagnification of per- and polyfluoroalkyl substances (PFAS). LEFT: PFAS bioaccumulation within an eel across its life stages, with contaminant concentrations increasing over time as the eel continues to take up PFAS from its environment and prey. RIGHT: PFAS biomagnification is illustrated through an aquatic food-web sequence, showing concentrations increasing from mayflies to small fish, larger predatory fish, and ultimately anglers/fish consumers Drawing by Carys.

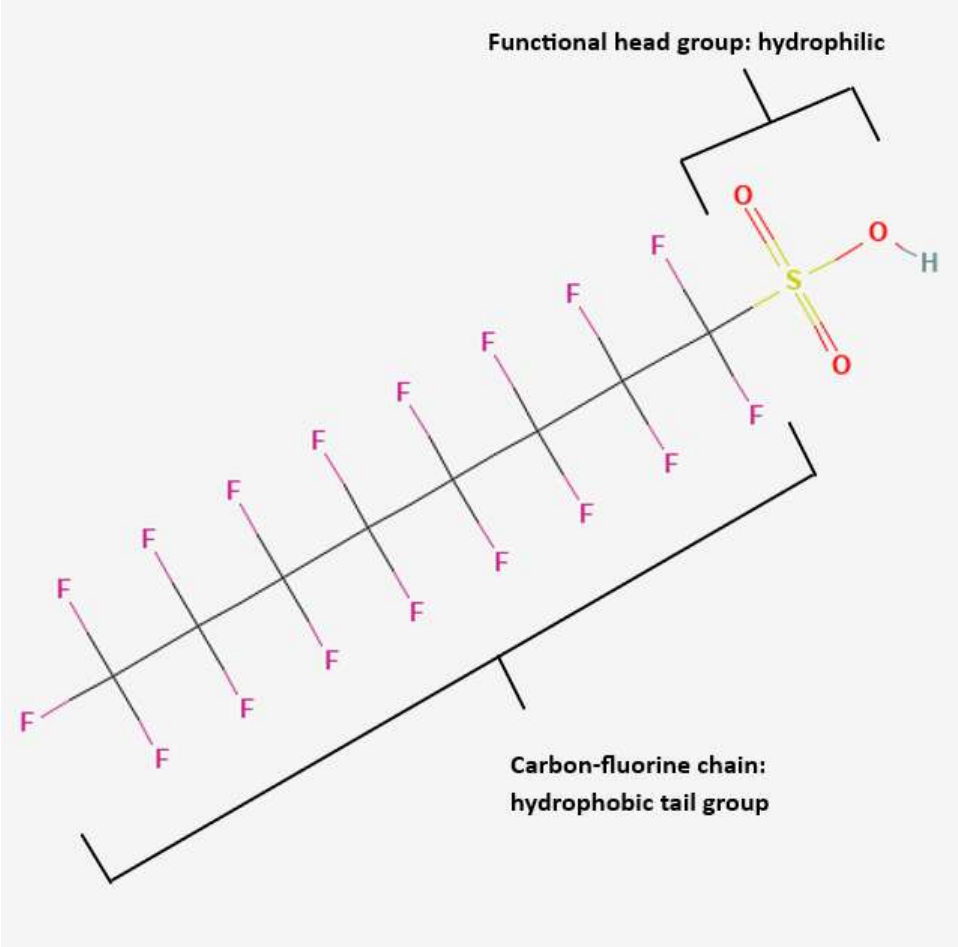

Figure 2: Perfluorooctanesulfonic acid (PFOS) structure illustrating the typical PFAS architecture, consisting of a carbon–fluorine chain and a polar functional head group (National Centre for Biotechnology Information (NCBI), 2008). The head groups are generally hydrophilic, whereas the perfluorinated tail (carbon–fluorine chain) is hydrophobic. Image retrieved from the PubChem database, with annotations added by Carys.

Fish species are differentially exposed to PFAS, both in terms of concentration and composition, depending on their habitat preferences and feeding mechanisms. Higher concentrations of PFAS have been observed in pelagic feeders that feed in the water column, compared to benthic feeders, which feed on the bottom (Giari *et al.*, 2023; Adeogun *et al.*, 2024). This suggests that the aquatic zone where an organism feeds influences exposure to and subsequent uptake of PFAS. However, this is not always the case and Sands *et al.* (2024) reported higher PFAS concentrations and a broader suite of compounds in the benthic species *Ictalurus punctatus* (Rafinesque, 1818) (Channel catfish), suggesting that trophic position may

exert a stronger influence on PFAS accumulation than the feeding zone alone. Interestingly, no significant difference in bioaccumulation has been observed between marine and freshwater fish (Jonker, 2024). Although Zafeiraki *et al.* (2019) did find lower levels of PFAS in eels collected at the mouth of highly polluted rivers compared to those from further inland.

Literature on the effect of PFAS on the health of aquatic organisms is often contradictory. Whilst PFAS exposure is often linked with decreased body size in fish, many studies have also reported no measurable change in body size or condition, with some even reporting body size to increase with PFAS concentration in fish tissue (Cara *et al.*, 2022; Hedgespeth *et al.*, 2023; Mehdi *et al.*, 2024). However, evidence consistently suggests that in fish, high PFAS body burdens are linked with underdeveloped swim bladders, crooked body axes, eye deficiencies, and increased incidence of liver cancer, all of which could have consequences for survival and reproductive success (Ji *et al.*, 2008; Benninghoff *et al.*, 2012; Hamed *et al.*, 2024; Mehdi *et al.*, 2024). This highlights that the wildlife health consequences of elevated PFAS levels, both at an individual and population scale, are not well understood.

Human Health and Exposure

PFAS exposure has been associated with a broad range of adverse health effects in humans (Hamed *et al.*, 2024). Increased cancer incidence, particularly liver and kidney cancers, has been linked to PFAS exposure in highly exposed populations (Benninghoff *et al.*, 2012; Steenland & Winquist, 2021; Tanghal *et al.*, 2026). The International Agency for Research on Cancer (IARC) assessed the carcinogenicity of PFOA and PFOS to humans in 2023, concluding that PFOA is carcinogenic whilst PFOS is possibly carcinogenic (Zahm *et al.*, 2024). However, in humans, the strongest and most consistent evidence relates to immune suppression and endocrine disruption. Reduced antibody responses to vaccines and other markers of impaired immune function have been repeatedly associated with PFAS exposure

(Sunderland *et al.*, 2019; Liang *et al.*, 2022; DeWitt *et al.*, 2025). Endocrine-related effects are also widely reported, including altered thyroid hormone regulation and impaired foetal development (Stein *et al.*, 2014; Gaillard *et al.*, 2024). Several recent epidemiological studies have also reported links between PFAS exposure and higher low-density lipoprotein cholesterol, which is associated with increased cardiovascular disease risk in humans (Haug *et al.*, 2023; Song *et al.*, 2025). Vulnerability to PFAS-associated impacts appears greatest among infants, children and pregnant people (Sunderland *et al.*, 2019; MAO *et al.*, 2024).

Non-occupational human exposure to PFAS appears to be primarily through consumption of either contaminated water or contaminated aquatic organisms (Chen *et al.*, 2023). Regular consumption of PFAS-contaminated fish could mean that a person's daily exposure to PFAS exceeds recommended intake levels, and even low levels of PFAS in fish can contribute to cumulative body burdens resulting in higher risk of adverse health impacts (Barbo *et al.*, 2023; Polychronidou & Nag, 2025). To mitigate the risks of exposure via seafood consumption, some studies have recommended that people boil seafood, or keep organisms alive in a clean bucket of water for a period prior to cooking (Wang *et al.*, 2020). Cooking methods vary in their success at removing PFAS from aquatic organisms. Boiling or frying seafood can decrease PFAS efficiently by facilitating the release of PFAS into the cooking medium (water or oil) but baking increases PFAS concentrations of PFBA and PFOS (Chen *et al.*, 2023). However, it is unlikely that PFAS can be entirely eliminated by cooking and in fact some studies suggest that cooking seafood is entirely ineffective at reducing PFAS exposure (Taylor *et al.*, 2019). Avoiding consumption of offal may reduce exposure as research has indicated that PFAS accumulates in the fish liver and kidneys whilst fish muscle contains much lower levels of PFAS (George *et al.*, 2023). However, concentrations in fish muscle can still exceed human consumption guidelines (Soudani *et al.*, 2024). The argument has been made that the benefits of fish consumption could outweigh PFAS-associated health risks; few studies account for this, however, those that do are not supportive of this claim (Wikström *et al.*, 2020; Hamade, 2024).

Significance of PFAS knowledge gap in New Zealand context

Despite extensive concern over PFAS contamination in waterways globally and the associated threats to human and ecological health, they have had limited attention in New Zealand (Coakley *et al.*, 2018; Ankley *et al.*, 2021; Stockin *et al.*, 2021; Lenka *et al.*, 2022; Close *et al.*, 2023). Although PFAS have been detected in several New Zealand waterbodies and in marine mammals, to the best of our knowledge only one study, limited to two eel samples and one PFAS has examined potential PFAS contamination in freshwater fish(Shaw & Kingi-Hudson *et al*., 2019; Stockin *et al.*, 2021; Close *et al.*, 2023).

Anguilla anguilla (L., 1758) (European eel) is one of few species that shows consistently high levels of PFAS across ecosystems and countries (Zafeiraki *et al.*, 2019; Teunen *et al.*, 2021; Cara *et al.*, 2022). It has been suggested that European eels may have a particularly high potential for accumulating persistent organic pollutants, including PFAS, due to their predatory nature and long life, during which they remain in the same location until they migrate (Barry *et al.*, 2015). This has potential implications for New Zealand's own *Anguilla* species – *Anguilla dieffenbachii* Gray 1842 (New Zealand longfin eel) and *Anguilla australis* (Richardson, 1841) (short-finned eel), collectively known by māori as tuna. Tuna share many behavioural and morphological characteristics with European eel meaning it is likely that PFAS could bioaccumulate in tuna, although this has not been studied.(Feunteun *et al.*, 2003; Cresci, 2020). The extremely long life spans of longfin eels (35 – 50 years, up to 100, (Chisnall & Hicks, 1993)) presumably places them at greater risk of accumulation.

Recreational fishing holds significant commercial, cultural, and social value for both indigenous and non-indigenous peoples of Aotearoa New Zealand. Consumption of freshwater fish in Aotearoa New Zealand, by indigenous Māori populations and amateur anglers, is a major recreational and cultural activity. Additionally, consumption of fish and seafood by Māori is approximately twice that of the general population. Furthermore, eel species (*Anguilla* spp.) that have been found in international studies to accumulate very high

PFAS concentrations are regularly consumed and hold significant cultural and spiritual importance for Māori (Zafeiraki *et al.*, 2019; Parai, 2021; Teunen *et al.*, 2021). Understanding PFAS contamination in aquatic food webs and its implications for the health of indigenous communities, the environment, and the general population is critical for evaluating health risks and safeguarding cultural values.

Overall, the literature reveals inconsistent findings, suggesting that the biomagnification or bioaccumulation patterns of PFAS are intricate and influenced by various factors, including biotransformation, ecological characteristics, and the physicochemical properties of PFAS highlighting the need for local studies to gain a true understanding of local impact.

Research Objectives and Thesis Outline

This research aims to address some of the key knowledge gaps surrounding PFAS in freshwater ecosystems in New Zealand. Specifically, I will investigate the occurrence, concentration, and composition of 75 PFAS in six freshwater fish species from populations in the lower North Island, from both “pristine” areas and areas downstream of potential contamination sources where fish are harvested for consumption. I will also examine inter-species differences in PFAS accumulation and discuss whether the consumption of aquatic organisms represents a potential pathway for human exposure in New Zealand. Additionally, I will explo methods that could inform future monitoring and assessment of PFAS in New Zealand’s freshwater fisheries.

In Chapter 1 I have contextualised PFAS contamination by reviewing international findings which may be relevant in New Zealand, specifically as they relate to eels which are a culturally significant food source for tangata whenua and, owing to their life history traits that facilitate bioaccumulation, may present pathways to human exposure.

In the following Chapters I will investigate the occurrence and composition of PFAS in New Zealand freshwater fish, the differences between fish species, and assess whether PFAS

concentrations in New Zealand fish may present a human health risk. Additionally, to further the potential for wide-scale PFAS monitoring in fish, both in New Zealand and elsewhere, I will explore using fish fins as a non-lethal method of tissue collection for estimating PFAS body burdens in fish. This is important to investigate because many New Zealand fish species are threatened or endangered, and fin-clipping is a non-lethal method used for assessing contaminant burdens in fish (Sanderson *et al.*, 2009; Jovičić *et al.*, 2023).

In the final Chapter I will provide a synthesis of my findings and discuss the limitations of the research I have undertaken and propose a direction for future research into PFAS and their impact on New Zealand's freshwater fish and those that consume them.

2 Chapter 2: Materials and Methods

Fish Sample Collection

Fish samples were obtained in two ways. Firstly, in July 2025, I collected fish at potentially contaminated sites historically used for food harvesting, using an EFM300 Electric Fishing Machine at shallow, flowing-water sites and using fyke and minnow traps at deep, still-water sites. These sites were located in three urban waterways (sites D- F) and one rural waterway downstream of an Air Force base (C - Lower) (**Table 1**, **Fig. 3**). Species captured at these sites were the native short-fin eel (short-fin eel, n = 18), *Gobiomorphus* spp. (bully, n = 8), and introduced *Perca fluviatilis* L, 1758 (European perch, n = 6) and *Gambusia affinis* (Baird & Girard 1854) (mosquito fish, n = 10). Also caught was a *Cyprinus carpio* L, 1758 (common carp, n = 1).

Fish were euthanised using Aqui-S ®, an aquatic anaesthetic derived from clove oil, following our ethics approved protocol (Massey Animal Ethics Approval No. AEC 25/28). Fish were dissected in the field with a fresh sample blade for each fish while wearing powder-free nitrile gloves. Liver samples were collected from all large fish and fin samples were collected from eels at site C - lower and eels and perch collected at site E. Livers and fins from larger fish were placed in PFAS-free polyethylene bags. The fish were then respectfully buried. *Gambusia* and bullies were too small to sample adequate tissue in the field, so the whole fish was collected as above. All sample bags were placed in cooler bins, transported to the laboratory, and frozen at - 20 °C within 6 hours. Whole fish samples were defrosted at room temperature and dissected, using a fresh blade for each fish. Samples were placed in polyethylene vials and refrozen at - 20 °C.

Secondly, in May 2025 I asked four local freshwater angler groups across the lower North Island of New Zealand (Auckland Flyfishing Anglers Club Inc, Kapiti Fly Fishing Club, Manawatū Freshwater Angler's Club Inc., Wellington Flyfishers Club Inc.) to supply liver samples from recreationally caught fish (Massey Human Ethics Approval No. 4000030385).

Sampling kits (PFAS-free sampling bags, powder-free nitrile gloves, handling instructions) were prepared and distributed to the fishing clubs. This collection method yielded fish liver samples from three popular fishing sites between May and July 2025 (Sites A, B, C Upper). These sites are all on relatively pristine, larger rivers of the central North Island. The samples were received as livers already removed and frozen in polyethylene bags and comprised mostly adult *Oncorhynchus mykiss* (Walbaum, 1792) (rainbow trout, n = 19) and one *Salmo trutta* L, 1758 (brown trout).

This yielded a total of 64 liver and 17 fin tissue samples from six fish species collected at seven North Island sites. Of the collected species, eel, perch and both species of trout are all consumed by fisherman and tangata whenua.

Site anonymity was guaranteed to local fishing clubs and food gatherers. Therefore, only general site locations and descriptions, and the species caught are provided here (**Table 1, Fig. 3**).

I obtained the required biosecurity clearances and documentation to export the samples to the Australian Laboratory for Emerging contaminants where I conducted PFAS extraction and analysis. The samples were transported frozen.

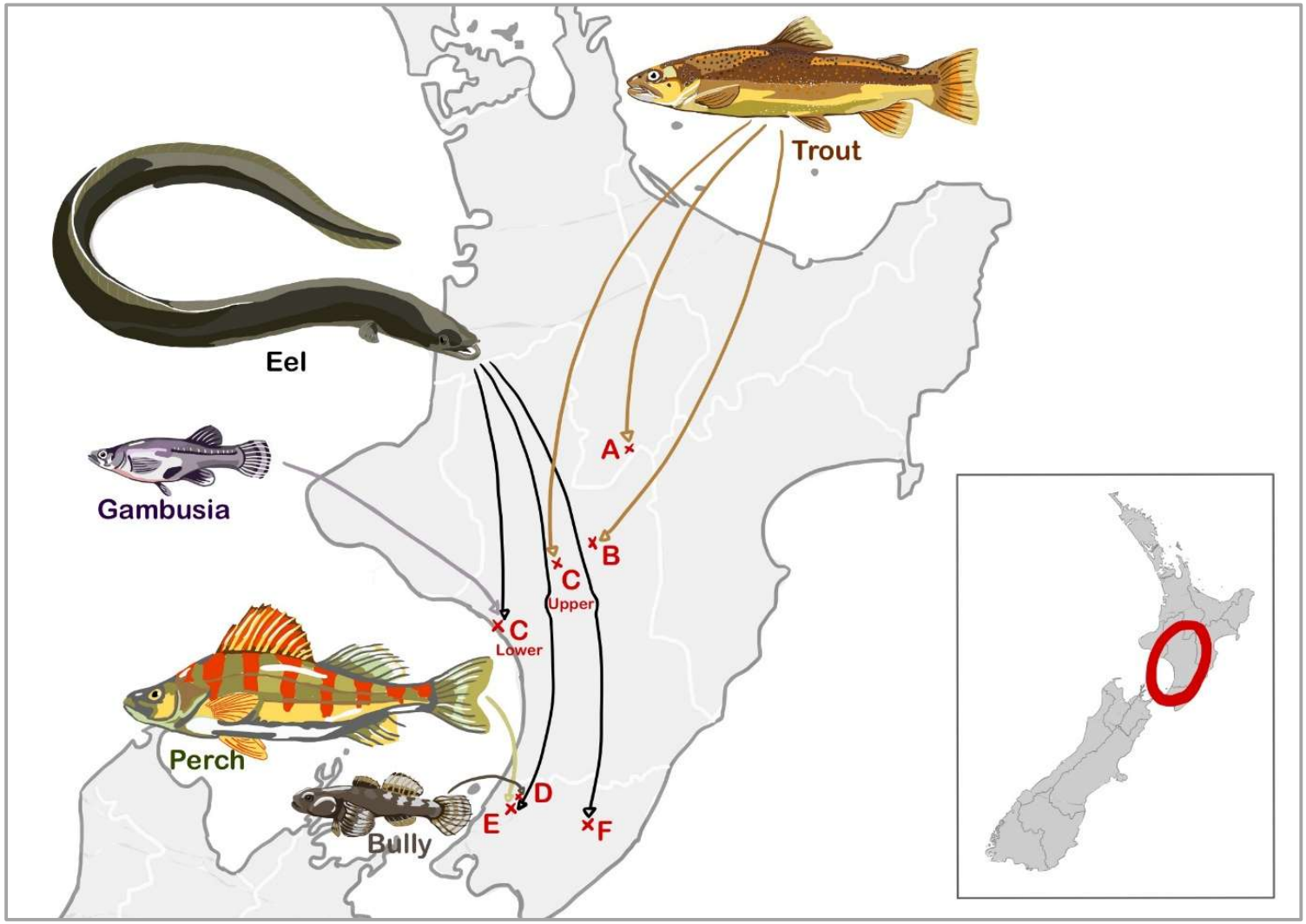

Figure 3. Seven sites (A-F) where fish species were caught for tissue analysis of per- and polyfluoroalkyl substances (PFAS) in the North Island of New Zealand between April and July 2025.

Table 1. Site descriptions and list of fish species caught at each site the lower North Island of New Zealand between April and July 2025. Site coordinates withheld to preserve anonymity.

Site Name	Description	Species	Common Name	*n*
Site A	Popular trout fishing site in a national park, upper catchment, pristine environment.	*Oncorhynchus mykiss*	Rainbow Trout	5
Site B	Popular trout fishing site, rural area, upper catchment	*Oncorhynchus mykiss*	Rainbow Trout	11
		Salmo trutta	Brown trout	1*
Site C Upper	Popular trout fishing site, rural area, upper catchment, same catchment as Site C Lower	*Oncorhynchus mykiss*	Rainbow Trout	3
Site C Lower	Historic mahinga kai (food gathering) site, downstream of a military base, lower catchment, same catchment as Site C Upper	*Anguilla australis*	Short-finned Eel	9
		Gambusia affinis	Mosquito fish/ Gambusia	10
Site D	Located in the lower reach of a large, urbanised river catchment.	*Anguilla sp.*	Eel sp. (juvenile)	1
		Gobiomorphus sp.	Bully	9
Site E	Small urban stream with multiple stormwater inputs. Occasionally used for food gathering by the local community. A small wastewater treatment plant discharges upstream of this site.	*Anguilla australis*	Short-finned Eel	4
		Perca fluviatilis	European perch	6
		Cyprinus carpio	Common Carp	1
Site F	Urban pond fed by stormwater and urban river. Samples collected from the main channel of the river just upstream of the pond and the pond. Occasionally used for food gathering by the local community	*Anguilla australis*	Short-finned Eel	4
TOTAL				64

PFAS Extraction: Chemicals and Standards

Hypergrade methanol (MeOH, >99.9 %), acetonitrile (ACN, 99.9 %), isopropanol (IPA, 99.9%) used in the extraction process were purchased from Sigma-Aldrich (New South Wales, Australia). A Milli-Q reference A+ system with reverse osmosis water was used to procure ultrapure water for this analysis (18.2 Ω, <5 ppm TOC, Merck, New South Wales, Australia).

Seventy-five PFAS were quantified using a 100 ng/mL stock solution made from 23 isotopically mass-labelled PFAS internal standards and hypergrade methanol. Control samples used a native stock solution which contained a combination of 75 native PFAS solutions and hypergrade methanol. Internal standards and native solutions were purchased from Wellington Laboratories (Ontario, Canada).

All consumables were tested for contamination prior to use by performing the extraction method without the addition of tissue.

PFAS Extraction from Fish Tissue

Samples were stored frozen at −30°C and thawed at room temperature for ~30 min prior to extraction. Approximately 100 mg (wet weight) of fish tissue was required for extraction. Liver samples were subsampled and extracted in triplicate, whereas fin samples contained sufficient tissue for only a single replicate. For the small whole fish (e.g., gambusia), internal tissues were pooled to obtain adequate material; these are referred to as mixed tissue samples. Sample weights were recorded using a Mettler Toledo XSR205DlJ analytical balance.

Replicates then received 49.5 µL of mass-labelled PFAS internal standard, 49.5 µL MeOH, 1 mL IPA and approximately 30 ceramic beads for homogenization purposes. Samples were homogenized before being frozen at −30°C for 10 min to allow lipids to separate out. Following cooling, samples were vortexed for 10 s and centrifuged at 0°C for 10 min at 4000 rpm. From each sample, 800 µL of supernatant was extracted from the top of each

sample and transferred to 15 mL centrifuge tubes containing 50 mg C18 sorbent. One mL of ACN was added, and tubes were vortexed for 10 s before freezing at −30°C for 10 min. Samples were centrifuged again at 0°C for 10 min, and an additional 900 µL of supernatant was transferred to the same tubes. This two-step extraction achieved 96% recovery. This was verified by spiking liver samples with a known concentrations of native PFAS solution and measuring the amount recovered.

Extracts were frozen at −20°C for 10 min and then centrifuged to allow the C18 to settle. 1.5 mL of extract was then transferred to new 15 mL tubes, and 40 µL Milli-Q water was added. Extracts were evaporated to 40 µL at room temperature under a gentle flow of nitrogen gas. After evaporation, 160 µL MeOH was added, and samples were vortexed for 10 s, frozen at −20°C for 10 min, and centrifuged at 4000 rpm for 10 min. Finally, 100 µL of each extract was transferred to 250 µL polypropylene vials for LC-MS/MS analysis.

Instrumental Analysis

LC-QQQ (liquid chromatography-tandem quadrupole mass spectrometry) analysis was performed using an Agilent 1290 Infinity II liquid chromatography system (LC), coupled with an Agilent 6495C tandem mass spectrometer (MS/MS) equipped with Agilent Jet Stream negative electrospray ionisation (AJS ESI-) (Agilent Technologies, USA) to detect and quantify target PFAS in all samples. The same instrument column was used for all samples across all batches. The method used in this study followed the LC-MS/MS method set out in Partington *et al.* (2024), which is capable of detecting 75 different PFAS, from nine different subgroups. PFAS are placed into subgroups based on their functional head group (Buck *et al.*, 2011). See **Appendix A** for all compounds analysed for and their associated subgroup.

Eleven calibration samples were produced via serial dilution of PFAS calibration stock solution in solvent to quantify PFAS in samples. The concentrations of the calibration samples ranged from 0.01 to 50 ng/mL, but several samples fell outside of this calibration range so

they were diluted with MeOH until they read within the range. The dilution factor varied between individual samples requiring this treatment. Quantification was undertaken using MassHunter QQQ quantitative analysis software (version 10.1, Agilent Technologies, USA).

Following the censoring method applied by Marchiandi *et al.* (2024), an MRL (method reporting limit) was calculated for each PFAS based on the lower of the method blank and the lowest calibration level that had a signal-to-noise ratio (S/N) greater than 10. If a measurement had an S/N less than three, it was considered a non-detect (ND) and treated as zero in subsequent analysis. Measurements with an S/N above three but lower than 10 were classified as detections but reported as '< MRL'.

Quality Control/Quality Assurance (QA/QCs)

Samples were run in batches of 20 alongside two method blanks and two laboratory control samples, (one of each with fish tissue, one of each without). Controls and blanks were treated the same as samples, except for the addition of 49.5µL of native PFAS solution to laboratory controls added in place of 49.5µL of MeOH at the beginning of the extraction.

An initial non-injection was followed by an injection of MeOH to detect and remove instrument contamination. For most method blanks, analyte concentrations were either not detectable (ND) or below the MRL. In a few instances of minor contamination, the blank concentration was used instead of the MRL to censor results within the corresponding batches.

Prior to testing samples, the extraction method was trialled using control samples with native solution and fish liver samples subsampled from a large liver of a trout caught at Site A. Several solvents and combinations were tested and those used in the final extraction method were found to result in the best compound recovery and the least compound degradation.

Statistical Analysis

Statistical analyses were performed in R version 4.3.3 (R Core Team, 2024). Differences in PFAS concentration between species and sites were examined with Analysis of Variance (ANOVA) and Tukey's HSD *post-hoc* comparisons.

Differences in the composition of PFAS subgroup among species and sites were investigated with model-based analysis of multivariate abundance on log-transformed concentrations (manylm) using the *mvabund* package, and cor.type=R, as all subgroups but diPAP were correlated (Wang *et al.*, 2012). Univariate permutation-based F-tests, adjusted using a stepdown resampling procedure, were used to identify significant differences in PFAS subgroup between species. An NMDS ordination, using a Bray-Curtis distance on log-transformed concentrations of PFAS subgroups, was conducted with the *vegan* package to visualise differences between species and sites in multivariate space (Oksanen, 2019). The relationship between fin and liver perfluorooctanesulfonic acid (PFOS) concentrations was analysed using $\log_{10}$-transformed linear regression in base R.

3 Chapter 3: Results

Of the 75 PFAS assessed, 46 compounds were detected above MRL in at least one sample. The detected compounds spanned six chemical subgroups: Fluorotelomer Carboxylic Acids (FTCAs), Fluorotelomer Sulfonic Acids (FTSAs), Perfluoroalkyl Sulfonamides (PASFs), Perfluorocarboxylic Acids (PFCAs), Perfluorosulfonic Acids (PFSAs), and diester Polyfluoroalkyl Phosphate Esters (diPAPs). The compounds Perfluorobutane sulfonamide (FBSA) and PFOS were detected in the majority of samples (87% and 84% respectively), and an additional four compounds were detected in at least 50% of samples (Perfluorodecanoic acid (PFDA), Perfluorotetradecanoic acid (PFTeDA), Perfluorotridecanoic acid (PFTrDA), and Perfluorododecanoic acid (PFDoDA)) (**Fig. 4**).

PFAS Concentrations

Distinct interspecific differences in PFAS concentration were present ($F_{4,57}$= 4.92, P< 0.002). Tukey post-hoc tests showed that eels had significantly higher ∑PFAS concentrations than bullies and trout (all $P < .01$), while other pairwise differences were non-significant. Eels had the highest mean concentrations of ∑PFAS (mean = 1368 ng/g ww), over twice as high as *Gambusia* (mean ∑PFAS 643 ng/g ww), 35 times higher than perch (mean ∑PFAS 39 ng/g ww) and over 1000 times higher than trout and bullies (mean ∑PFAS 1.1 ng/g ww, and 2.25 ng/g ww, respectively) (**Table 2, Fig. 5**). Differences in ∑PFAS were primarily a result of PFOS concentration. Eel livers had the highest PFOS concentration, ranging from 4.9 ng/g in a juvenile caught at Site D to 3,867 ng/g in an adult at Site C.

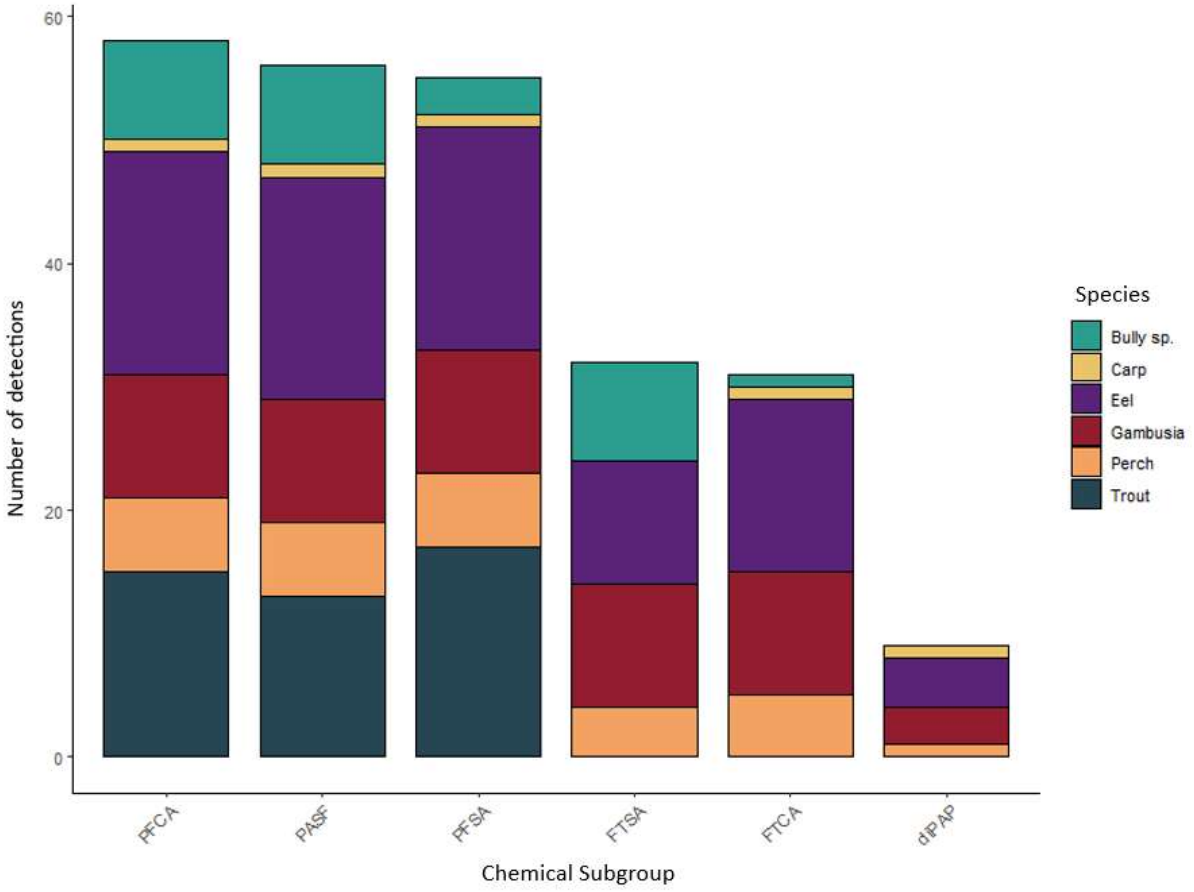

Figure 4: Number of detections for per- and polyfluoroalkyl (PFAS) chemical subgroups: perfluoroalkyl carboxylic acids (PFCA), perfluoroalkyl sulfonyl fluorides (PASF), perfluoroalkyl sulfonic acids (PFSA), fluorotelomer sulfonic acids (FTSA), fluorotelomer carboxylic acids (FTCA), and polyfluoroalkyl phosphate diesters (diPAP) across six species of freshwater fish collected at seven sites in the North Island, New Zealand, between April and July 2025.

Table 2: Total concentrations of per- and polyfluoroalkyl substances (PFAS), as well as perfluorooctanesulfonic acid (PFOS), in fish liver and mixed-tissue samples, expressed as nanograms of PFAS per gram of wet weight (ng/g ww), from six species of freshwater fish collected at seven sites in the North Island of New Zealand between April and July 2025.

Site	Species	Common Name	Tissue type	Mean ∑PFAS	Range ∑PFAS	Mean PFOS	Range PFOS	*n*
Site A	*Oncorhynchus mykiss*	Rainbow Trout	Liver	0.9	0.2 – 1.5	0.9	0.88 – 0.90	5
Site B	*Oncorhynchus mykiss*	Rainbow Trout	Liver	1.5	1 – 4	0.9	0.85 – 1.1	11
	Salmo trutta	Brown Trout	Liver					1*
Site C Upper	*Oncorhynchus mykiss*	Rainbow Trout	Liver	0.7	0 – 1.3	0.7	0 – 1.3	3
Site C Lower	*Anguilla australis*	Short-finned Eel	Liver	2693	996 - 7178	1636	878 - 3867	9
	Gambusia affinis	Mosquito fish	Mixed tissue	643	314 - 1216	607	300 - 1140	10
Site D	*Anguilla sp.*	Eel (juvenile)	Liver	6.4	-	4.9	-	1
	Gobiomorphus sp.	Bully	Mixed tissue	2.3	0.5 - 8	0.33	0 – 0.9	9
Site E	*Anguilla australis*	Short-finned Eel	Liver	75	61 - 86	52	44 - 55	4
	Perca fluviatilis	European Perch	Liver	39	29 - 49	25	15 - 35	6
	Cyprinus carpio	Common Carp	Liver	44	-	33	-	1
Site F	*Anguilla australis*	Short-finned Eel	Liver	23	5 – 37	13	0 - 18	4

**For analysis purposes Brown Trout and Rainbow Trout have been combined into one 'Trout' category*

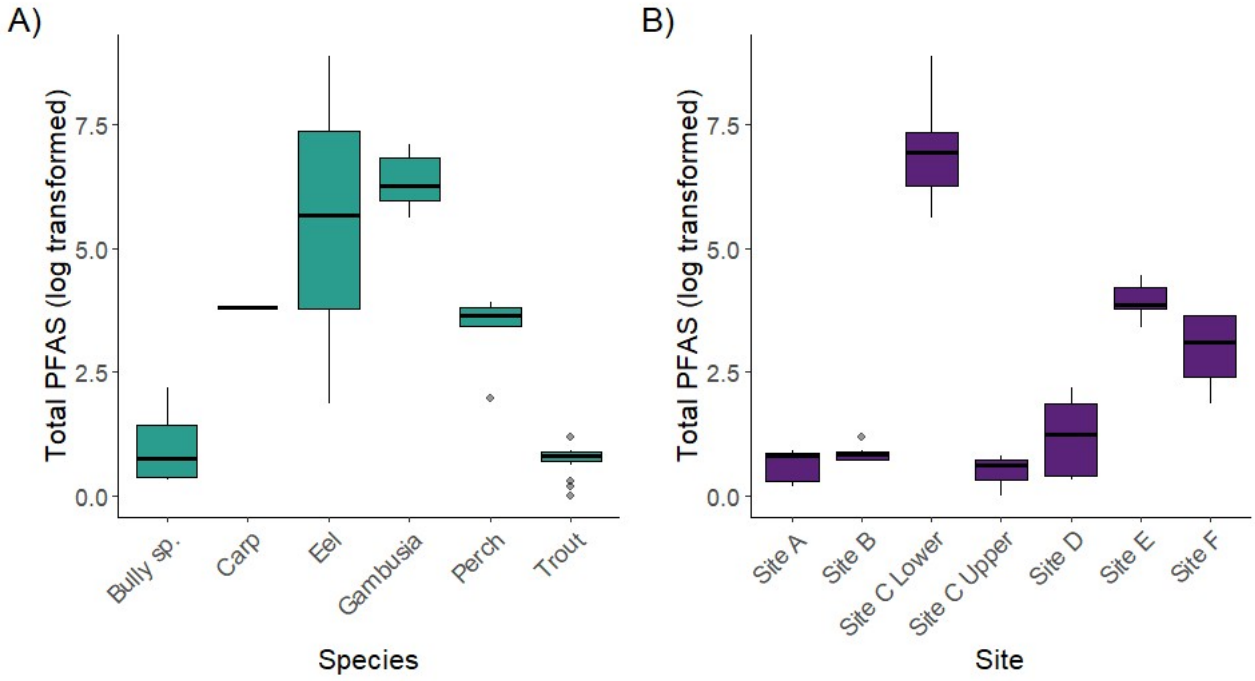

Figure 5. Boxplots of ∑PFAS concentrations in freshwater fish collected across seven sites in the North Island, New Zealand between April and July 2025: (A) ∑PFAS concentrations by species sampled, (B) ∑PFAS concentrations by sampling site. Boxplots display the median (horizontal line), interquartile range (IQR; box), and whiskers extending to 1.5 × IQR. Points beyond the whiskers represent statistical outliers. Concentrations are shown on a log-transformed scale to aid visual comparison across orders of magnitude.

Not surprisingly, given the dissimilar habitats, the types of fish collected at the sites differed. The potentially contaminated urban sites were dominated by *Anguilla*, *Gobiomorphus* and *Gambusia* whereas at the more pristine, larger rivers only trout were collected. This complicates the assessment of species versus site impacts on PFAS concentrations. Thus, although I found differences in PFAS concentrations among fish species and sites, it is difficult to know if the differences are due to differences in species response or if it was simply that only certain species were present in the contaminated sites.

PFAS Composition

Overall contaminant profiles were different between species ($F_{4,57}$ = 187.06, P > 0.01) and sites ($F_{6,51}$ = 191.08, P > 0.01) with no interaction between species and sites ($F_{24,51}$ =11.61, P = 0.995). FTCAs, FTSAs, PASFs, PFCAs, and PFSAs all contributed to differences in profiles, but diPAPs did not differ ($F_{5,57}$ = 1.667, P > 0.2) (**Fig. 6**). Across most species, PFSAs represented the largest proportion of the contaminant profile. L-PFOS was the main compound responsible for the abundance of PFSA, accounting for between 96 and 100 % of PFSA.

In an NMDS ordination of species contaminant profiles (**Fig. 7**), all eels (except one) appear to the lower right of axis one and are associated with greater PFSA. Bullies are to the left of axis one and are associated with FTSA and PASF, and trout are spread across axis one, but to the bottom of axis one with lower amounts of PFCA and PFES.

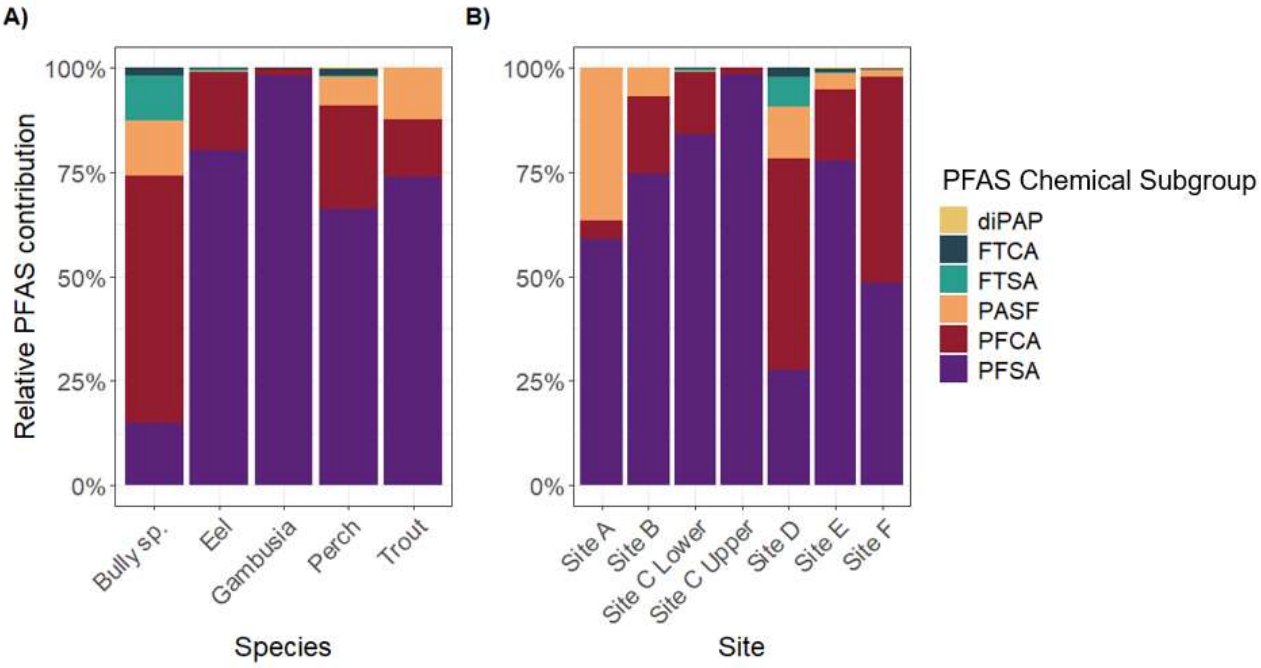

Fig 6. Relative abundance of PFAS chemical subgroup recorded in five species of freshwater fish collected at seven sites in the North Island, New Zealand, between April and July 2025: (A) composition chemical subgroups of PFAS within each fish species; (B) composition chemical subgroups of PFAS at each sampling site. The abbreviations of chemical subgroups of PFAS are explained in Figure 4.

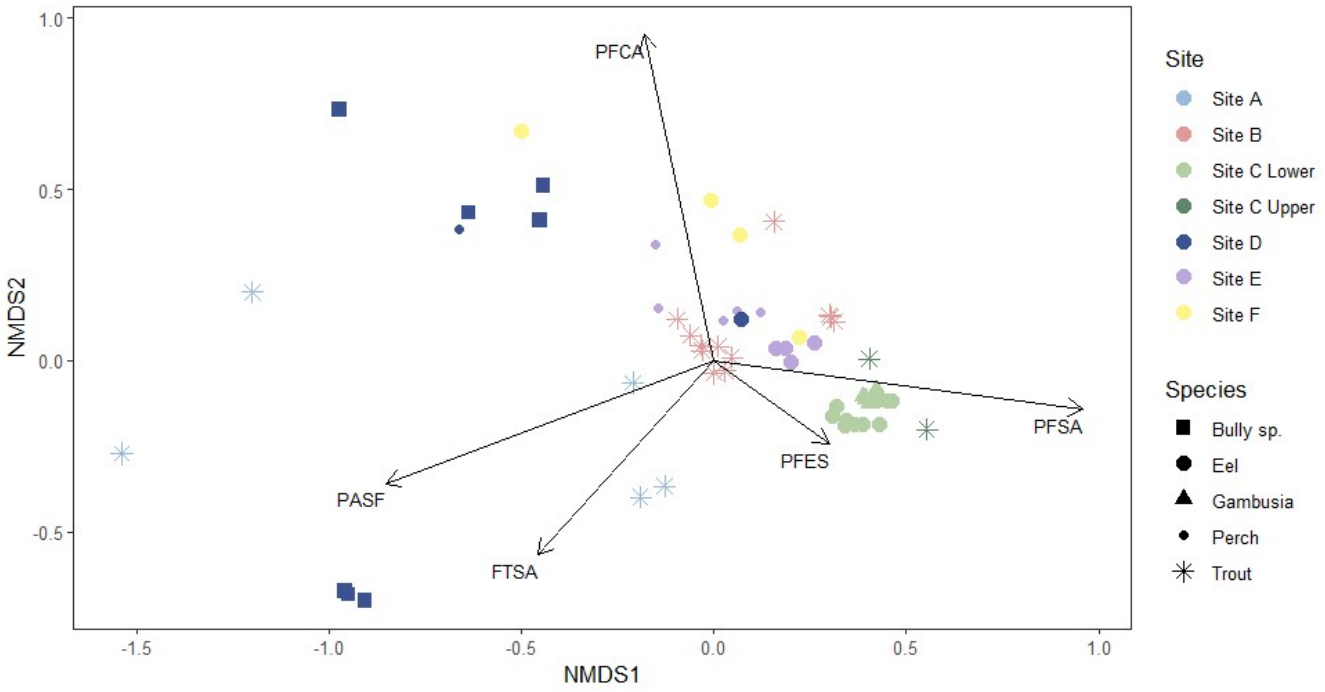

Figure 7. NMDS ordination of PFAS profiles for five species of freshwater fish collected at seven sites in the North Island, New Zealand, between April and July 2025. Vectors represent PFAS chemical subgroups correlated with axes in the ordination. Stress value = 0.06. PFAS chemical subgroups abbreviations are explained in Figure 4.

Links between liver and fin PFAS concentrations

There was a strong positive log-linear relationship between PFOS concentrations in the fish fins and liver ($F_{1,6}$ = 193.9, P < 0.001, r^2 = 0.92) (**Fig. 8**). Fin clipping is a non-lethal method for obtaining fish tissue and this finding provides promise for non-lethal tissue collection in the future to estimate PFOS burden in fish, including threatened and endangered species (Sanderson *et al.*, 2009; Jovičić *et al.*, 2023).

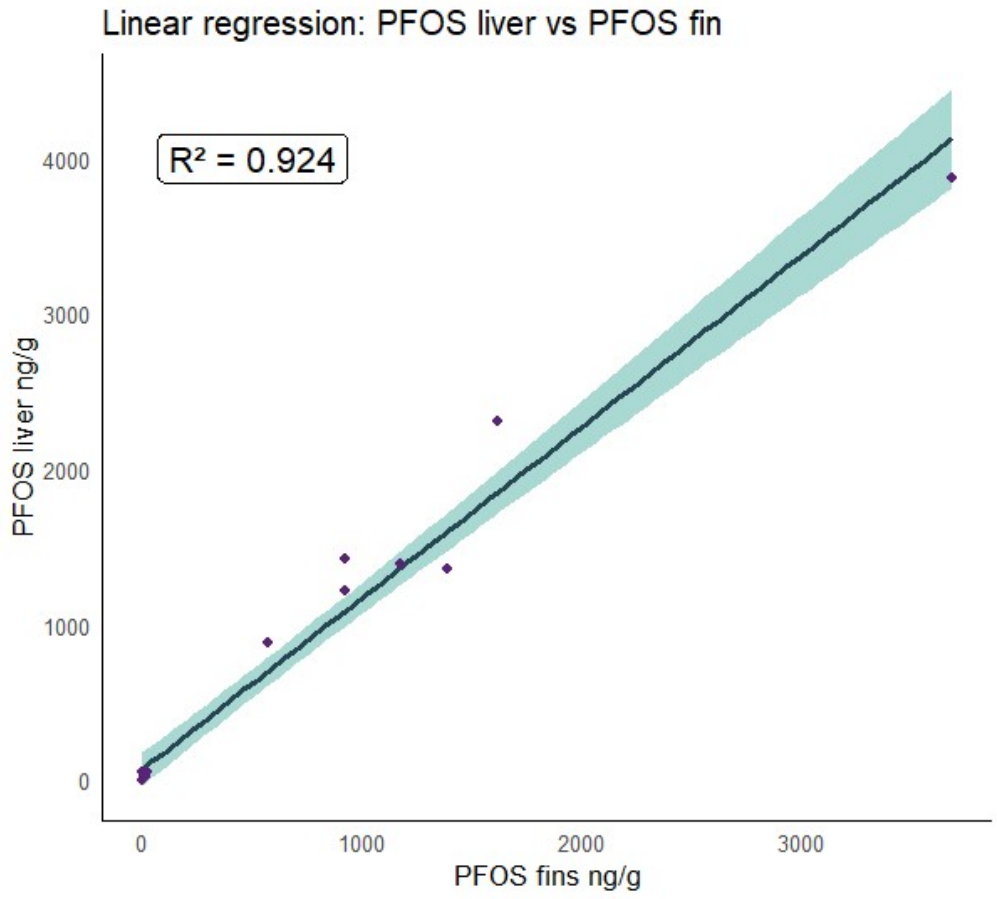

Figure 8. The PFOS concentrations in fish (N=17) livers as a function of PFOS concentrations in fins from those same fish classes recorded in eel and perch collected at three sites in the North Island, New Zealand, between April and July 2025. OLS regression, $P < 0.001$, $\log_{10}(\text{Liver}) = 0.74 + 0.78 \times \log_{10}(\text{Fin})$.

4 Chapter 4: Discussion

Per- and polyfluoroalkyl substance (PFAS) contamination is pervasive in freshwater ecosystems globally (Evich *et al.*, 2022). Despite growing international awareness of their persistence, mobility and toxicity, relatively little is known about the distribution and effects of PFAS in Aotearoa New Zealand's freshwater environments. In the following sections, I discuss the environmental patterns observed in this study, their implications for food safety, and offer insight into potential methodologies for future PFAS research in Aotearoa.

Concentrations

Differences in the PFAS concentrations between trout and the other species may reflect the habitat preferences of these species, as the latter were only found at the sites with potential for contamination. PFAS contamination is typically associated with urban streams, where eel, and *Gambusia* (but not trout) live, and where PFAS from industry and consumer products have more potential to contaminate waterbodies. Furthermore, one of our study sites has been identified as having been contaminated with PFAS from firefighting foam (Pattle Delamore Partners LTD, 2018). Relative to other PFAS, PFOS is less water-soluble, has stronger sorption to sediments and therefore accumulates more readily in sediment than the water column (Fagbayigbo *et al.*, 2022; Fatima *et al.*, 2025). This may explain some of the interspecific differences. Fish caught at sites with soft sediment substrate (C lower, E and F) had higher concentrations of PFOS than those caught in rocky-bottom waterways. Although, as mentioned above, the soft sediment sites are also urban streams with more potential for contamination. However, even within urban sites, short-fin eels, which are bottom feeders, had approximately twice the PFOS concentration (mean = 1636 ng/g) as the more pelagic feeding *Gambusia* (mean = 602 ng/g) (van der Oost *et al.*, 1996; Pyke, 2005). Differences in feeding behaviour could thus be exposing eels to higher levels of PFOS than *Gambusia*. More generally, fish species inhabiting waterways with more soft sediment may accumulate more PFAS than fish in waterbodies with gravel substrates. Whilst fish species and contamination levels were confounded, trout, who are water column feeders in rocky-bottom streams (Haury *et al.*, 1999), had the lowest mean PFOS concentrations of any species. My results and the wider literature, therefore, suggest that proximity to discharge point sources, mobility of species, substrate type, and species feeding behaviour all contribute to greater PFAS accumulation in some fish species compared to others, although more extensive sampling across a range of habitats and species will be needed to quantify this.

Contaminant Profiles

Species-level differences in contamination profile may reflect differences in fish life span because longer-lived species like eels (average lifespan of 18-23 years but up to 60 years) have more time to accumulate contaminants (Vardy *et al.*, 2026). Smaller fish species with shorter life-spans, such as the bullies and *Gambusia* (3-5 years and 1-2 years, respectively), will have lower legacy compounds compared to the longer-lived eels (McDowall, 1990). PFOS has been phased out of usage in New Zealand following the 2006 import ban, so this may be contributing to profile differences (Parliamentary Commissioner for the Environment, 2022). However, *Gambusia*, the shortest-lived species, did have the highest relative contribution of PFOS (99%). However, it should be noted that *Gambusia* were only collected at Site C, where water assessment by local authorities indicate PFOS is an ongoing issue from AFFF contamination (Pattle Delamore Partners LTD, 2018).

By site, most locations were PFSA dominated, except Site D and to a lesser extent Site F, where PFCA's made a substantial contribution to contaminant profile (**Fig. 6B**). Site D also exhibited a greater variety of compounds which is consistent with its position in the lower reaches of a large river. Lower reaches of major rivers tend to have more variety in compounds because they are receiving potentially more inputs from upstream reaches, (Bradley *et al.*, 2021; Kaushal *et al.*, 2023). Site D lies near the end of a catchment draining ~ 125 km^2, including two townships. These greater upstream influences likely explain the higher PFAS diversity in fish from Site D compared to sites A – C, E, and F, which are located higher the catchment or on low-order streams (order 1 – 2). This suggests that fish inhabiting the lower reaches of large, urbanised river catchments will have a more diverse contaminant profile.

Not all fish species occur at all sites (**Table 1**), so species differences may partially reflect site-specific contamination patterns and vice versa. Where multiple fish species occur at the same site, evidence for intrinsic species differences driving contamination profile is more compelling. At Site C where *Gambusia* and eels co-occur, PFSA's dominated both species

profiles, driven by PFOS. However, PFCAs made a substantially larger relative contribution in eels (20%) than in *Gambusia* (<5%; **Fig. 6A**). In eels, contribution of PFCAs was attributed mostly to the legacy compound PFOA whilst in *Gambusia* PFNA drove PFCA contribution. Longer lifespan in eels resulting in cumulative PFOA exposure, and biomagnification associated with their top-predator status in New Zealand ecosystems may explain this difference (Zafeiraki *et al.*, 2019; Teunen *et al.*, 2021). The resulting implication for fish consumers is that longer-lived species may pose greater health risks than shorter-lived species because their profiles are dominated by legacy compounds that are harmful to human and ecosystem health.

Global Context

Mean PFOS concentrations in this study are comparable to the highest mean values recorded in eels (**Table 3**) (Hoff *et al.*, 2005; Shaw & King-Hudson, 2019; Environment Protection Authority, 2022). *Anguilla anguilla*, the European eel, is one of a few species that show consistently high levels of PFAS across habitats and countries (Zafeiraki *et al.*, 2019; Teunen *et al.*, 2021; Cara *et al.*, 2022). My finding, that short-fin eels (*A. australis*) have higher ∑PFAS than cohabiting species, is consistent with the global literature on persistent organic pollutants (POP's) and *Anguilla* species. Mean PFOS concentrations in eels from urban sites in this study were comparable to those reported for *A. anguilla* in Germany and Italy, while mean concentrations at my most contaminated site exceeded those documented in Belgium, where the previous highest maximum concentrations have been recorded (Hoff *et al.*, 2005; Guhl *et al.*, 2014; Roland *et al.*, 2014; Giari *et al.*, 2023). Although the maximum PFOS value I recorded was lower than the international maxima, it still falls within the upper range of levels previously reported in eels and is the highest recorded value in eels in the Southern Hemisphere (**Table 3**).

Table 3: Recorded site-level concentrations of perfluorooctanesulfonic acid (PFOS) in *Anguilla* species, in nanogram per gram of wet weight (ng/g ww) of liver.

Country	Species	Mean PFOS per site	PFOS range	Reference
Italy	*Anguilla anguilla*	4.29	0.4 - 6.28	Giari *et al.* (2023)
Germany	*Anguilla anguilla*	22.9	Not Provided	Guhl *et al.* (2014)
		27.5		
		13.7		
		35.8		
		13.4		
		30.6		
		32		
		14.6		
		15.6		
		49		
		16.9		
		42.3		
		8.3		
Belgium	*Anguilla anguilla*	1387*	17.3 - 9031*	Hoff *et al.* (2005)
Belgium	*Anguilla anguilla*	230.1	Not provided	Roland *et al.* (2014)
		329.1	Not Provided	
		31.1	Not Provided	
New Zealand	***Anguilla australis***	**1637**	**878 - 3867**	**This study**
		10	**0 - 18**	
		50	**44 - 55**	
New Zealand	*Anguilla australis*	410	-	Shaw and King-Hudson (2019)
Australia	*Anguilla australis*	197	160 - 220	Environment Protection Authority (2022)
		9.6	3.9 - 14	

*Per site values and ranges not provided.

Human Health Risks

Short-fin eels are commonly consumed by indigenous populations (Māori) in Aotearoa, New Zealand, as a food resource and is a critical component of traditional social and cultural events (Parai, 2021). New Zealand also has commercial harvesting and exportation of three species of eel, one of which, the long finned eel *Anguilla dieffenbachia*, is classified as “at risk – declining" by the Department of Conservation and endangered by the IUCN (Beentjes, 2025). PFOS concentrations recorded in eel livers in this study are well above several consumption guidelines and therefore pose a health risk to consumers. Whilst other PFAS compounds, including several novel compounds, were also detected in eels, levels were low compared to PFOS. A lack of toxicological data and/or relevant guideline values makes it difficult to assess any potential risks and clearly needs further investigation (Fenton *et al.*, 2021).

Food Standards Australia and New Zealand (FSANZ) set a tolerable daily intake (TDI) for PFOS of 20 ng/kg bodyweight/day (Food Standards Australia New Zealand (FSANZ), 2016). Because consumption guidelines are typically set for muscle (fillet) rather than liver, and because liver acts in detoxification and often contains higher contaminant burdens than muscle, I estimated muscle concentrations from liver using published conversion factors (Roberts, 2012; Schrenk *et al.*, 2020; Great Lakes Consortium for Fish Consumption Advisories, 2025). Whilst these values were derived for fish, they are not necessarily specific to eels (Soerensen *et al.*, 2023). Based on my estimated fillet concentrations (range = 0.4 - 375 ng/g, mean = 80 ng/g) and an average body weight for a consumer of 70 kg, a single 150 g meal exceeded the TDI when assessed on a per-day basis. However, TDIs are intended for long-term average exposure, not single-meal assessments, so extrapolating over a year is challenging given the variation in how many meals of eel an individual might have. There is no maximum single meal intake set by the FSANZ, however, MPI provided guidance suggesting that recreationally caught finfish containing more than 125 ng/g of PFOS should not be consumed by children or consumed more than once per month by adults (Ministry for the Environment & Ministry for Primary Industries, 2023). Half of my fillet equivalent tissue

samples would exceed this guideline. Note that the New Zealand PFOS consumption values in fish are based on estimated values due to a lack of local fish concentration data. The European Food Safety Authority (EFSA) have more conservative guidelines of a tolerable weekly intake (TWI) of 4.4 ng/kg of bodyweight and a maximum level allowed in *Anguilla* species intended as food of 35 ng/g (Schrenk *et al.*, 2020; European Commission, 2023). Based on an average consumption of 1kg of eel per year, my average estimated fillet concentration exceeds both the TWI and the maximum consumption level. Half of my individual samples also exceed these levels.

These elevated concentrations carry potentially serious consequences for consumer health and may also result in economic impacts for the commercial and customary eel fisheries such as reduced consumer confidence and loss of overseas market access. New Zealand's eel fishing sector exports approximately $4.3 million worth of eel products, primarily whole eel including the liver, each year (Seafood New Zealand, 2025). However, these impacts are not likely to be uniformly felt. Fish and seafood consumption is twice as high in Māori than it is in non-Māori, thus PFOS contamination will disproportionately affect Māori. Māori suffer disproportionately more from many illnesses and disease in New Zealand, and PFAS contamination of eel may be one, as yet investigate, contributor to this discrepancy (New Zealand Ministry of Health, 2012; New Zealand Ministry of Health, 2025). Eel is a spiritually significant and culturally important species to Māori and short-fin eel are regularly harvested and eaten as part of indigenous practice (Parai, 2021). In this context, PFAS pollution represents not only an environmental issue but a further legacy of colonisation, compounding existing inequities in access to safe, culturally appropriate food sources.

In 2019, Shaw and King-Hudson (2019) published a perspective in the New Zealand Medical Journal concluding that, based on very limited sampling (i.e. two eels), that there was "minimal risk of PFOS residues in eel to Māori consumers" (Shaw & King-Hudson, 2019 p. 1), However, the highest PFOS concentration they used as a worse-case scenario was approximately 10 x lower than the highest value recorded in my study. Their assessment was

also focused primarily on carcinogenic risk, despite the broader suite of adverse effects associated with PFAS exposure. In contrast, I conclude, that based on the values in this study, eel consumption could expose tangata whenua to PFOS, at levels that have been linked to adverse health outcomes.

Effectiveness of Fins as an estimate of PFAS body burden in fish

Fin clipping is a non-lethal method for obtaining fish tissue, and my findings support its promise for non-lethal tissue collection in the future to estimate PFOS burden in fish, including threatened and endangered species (Sanderson *et al.*, 2009; Jovičić *et al.*, 2023). Fin-based PFAS estimations may provide a valuable tool for ecological and environmental monitoring unrelated to human health, such as assessing long-term trends in PFAS contamination in at risk species, or understand spatial variation in exposure across catchments. However, because muscle is the tissue most commonly consumed, direct fin–muscle validation will likely be required before fins can be used with confidence in human-health risk assessments.

PFOS concentrations in fish fillets are usually around ten-fold lower than in liver (Soerensen *et al.*, 2023). The European Food Safety Authority's (EFSA) maximum consumption guideline for Anguillid species is 35 ng/g which if present in fillets could correspond to a liver concentration of 361 ng/g. My fin liver regression predicts this fillet concentration would occur when fin PFOS is approximately 209 ng/g in the fin, a level that was exceeded in more than half of my fin samples. Typically, although no published study has explored PFAS in fins specifically, fin clipping studies for other contaminants report that fin concentrations are substantially lower than muscle which is contrary to what my results suggest, however my muscle concentrations are estimates only (Rolfhus & Sandheinrich, 2008). Investigating the relationship between actual fillet concentrations and fin tissue, both in eels and other commonly consumed fish species, would facilitate larger scale non-lethal monitoring of fish populations for PFOS contamination. Based on estimated fillet

concentrations, my findings do suggest that fins may be a useful non-lethal proxy for PFOS monitoring in a human-health based context.

Future research

The effect of migration on the bioaccumulation of PFAS is understudied in aquatic species and given that many of New Zealand's fish are migratory, this is a clear knowledge gap. Also, the biological consequences for the native eel species are unknown. However, PFAS exposure does have the potential to adversely affect wildlife health and reproductive success (Mehdi *et al.*, 2024).

Multi-tissue testing across commonly consumed species would improve the accuracy of consumption guidance while incorporating water and sediment analyses would strengthen future research by clarifying contaminant pathways. These recommended future research directions also reflect key short comings of the present study. Due to the prohibitive cost of PFAS analysis in New Zealand, lack of PFAS-testing capability at New Zealand Universities, and short timeframes for obtaining permits required to export to Melbourne where I undertook tissue analysis, water and sediment testing were not feasible. To the best of my knowledge, PFAS tissue testing in New Zealand is currently available only at one laboratory, Assure Quality, and is prohibitively expensive, at $906 per sample (quote date February 2026). This is a key limitation to this and future research that could be solved by investment in affordable domestic analytical capacity and standardised sampling protocols.

5 Chapter 5: Conclusions

This study provides the first comprehensive assessment of per- and polyfluoroalkyl substances (PFAS) contamination in Aotearoa New Zealand's freshwater fish and highlights implications of perfluorooctanesulfonic acid (PFOS) bioaccumulation for both ecosystem health and food safety.

Despite limited reported use of PFAS in New Zealand (Ministry for the Environment, 2020; Lenka *et al.*, 2022), my results show clear evidence that PFAS are accumulating in New Zealand fish species at my study sites, with particularly elevated PFOS concentrations in the native Short-fin eel (*A. australis*). This has important consequences for tangata whenua who regularly harvest and consume this species.

In addition to characterising spatial and species-level patterns in PFAS occurrence, this work explored fin clipping as a practical sampling approach for PFOS monitoring. Fin clipping shows promise as a non-lethal approach for PFOS surveillance in fish populations, supporting both ecological monitoring and, with proper fin-muscle validation, human-health applications. Together, the ecological patterns and methodological insights presented here offer a foundation for evidence-informed monitoring and policy development in Aotearoa New Zealand.

Many studies that aim to understand the occurrence and movement of PFAS within ecosystems are small with limited sampling, which could be due to the politically charged nature of this contaminant and high cost of sampling. Whilst there is some understanding of the actual effects on the human body, how these contaminants might disproportionately affect different communities, with consideration to cultural differences in diet, is under researched. This unstable scientific landscape presents a challenge for decision makers, and literature does not currently address the functional needs of regulators. Further research on a larger scale in a more diverse range of ecosystems, climates, and cultures is required in order to fully understand the occurrence and impact of PFAS.

Overall, my results demonstrate a clear need for national-level action on PFAS contamination, with implications for public and ecosystem health, and the resilience of eel export markets. Critically, my research highlights the urgent need for broader contaminant monitoring in freshwater species that are central to indigenous food practices and recreational food gathering.

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7 Appendix A

List of 75 per and polyfluoroalkyl substances analysed for and the chemical subgroup they fall into.

Subgroup	Compound
AFFF	6:2 FTAB
	5:1:2 FTB
	5:3 FTB
	AP-FHxSA
	TAMP-FHxSA
diPAP	8:2 diPAP
	diSAmPAP
	6:2/8:2 diPAP
FTCA	7:3 FTCA
	5:3 FTCA
	3:3 FTCA
FTSA	8:2 FTSA
	6:2 FTSA
	4:2 FTSA
FTUCA	6:2 FTUCA
	8:2 FTUCA
	10:2 FTUCA
PASF	FHpSA
	EtFBSA
	EtFBSE
	FHxSA
	MeFBSE
	MeFBSA
	FPeSA
	FBSA
	FDSA
	8:8 PFPi
	6:8 PFPi
	6:6 PFPi
	MeFOSAA
	FOSAA
	EtFOSA
	EtFOSE
	MeFOSE
	MeFOSA

Subgroup	Compound
PFCA	Br-PFOA
	PFBA
	PFDA
	PFNA
	PFMPA
	PFHpA
	PFHxA
	PFMBA
	PFPeA
	PFODA
	PFHxDA
	PFTeDA
	PFTrDA
	PFDoDA
	PFUnDA
PFES	6:2 Cl-PFESA
	8:2 Cl-PFESA
	ADONA
	HFPO-DA
	NFDHA
	PFEESA
PFSA	8Cl-PFOS
	Br-PFOS
	Br-PFHxS
	PFECHS
	L-PFNS
	L-PFOS
	L-PFHpS
	L-PFOA
	L-PFHxS
	L-PFPeS
	L-PFBS
	L-PFPrS
	L-PFDS
	L-PFTrDS
	L-PFDoDS
	L-PFUdS