Bangor University

MASTERS BY RESEARCH

Determining the effects of anthropogenic drainage structures on inanga (Galaxias maculatus) in the lowland waterways of Hawke's Bay, New Zealand.

Halstead, Reece

Award date:
2022

Awarding institution:
Bangor University

Link to publication

PRIFYSGOL
BANGOR
UNIVERSITY

Determining the effects
of anthropogenic drainage structures
on inanga *(Galaxias maculatus)*
in the lowland waterways of
Hawke's Bay, New Zealand

Reece Halstead

A thesis submitted for the degree of
Master of Science by Research
in the School of Natural Sciences
at Bangor University

Supervisors: Dr Nigel Milner, Dr Nathalie Fenner, Daniel Fake

2022

Declaration to include in your thesis

I hereby declare that this thesis is the results of my own investigations, except where otherwise stated. All other sources are acknowledged by bibliographic references. This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless, as agreed by the University, for approved dual awards.

Yr wyf drwy hyn yn datgan mai canlyniad fy ymchwil fy hun yw'r thesis hwn, ac eithrio lle nodir yn wahanol. Caiff ffynonellau eraill eu cydnabod gan droednodiadau yn rhoi cyfeiriadau eglur. Nid yw sylwedd y gwaith hwn wedi cael ei dderbyn o'r blaen ar gyfer unrhyw radd, ac nid yw'n cael ei gyflwyno ar yr un pryd mewn ymgeisiaeth am unrhyw radd oni bai ei fod, fel y cytunwyd gan y Brifysgol, am gymwysterau deuol cymeradwy.

Executive Summary

The effects of human-induced river fragmentation on the migrations of diadromous fish is well documented in the literature, though research has focused on the effects of large-scale anthropogenic drainage structures on strong swimming salmonids, whilst the impacts of small-scale structures on weaker swimming fish is less well known. Despite being the most prevalent and widely distributed whitebait species of the Galaxiidae family, and having significant recreational, commercial and cultural importance in New Zealand, populations of the small-bodied inanga *(Galaxias maculatus)* are in decline. The amphidromous migrations of juvenile inanga require unrestricted passage into upstream habitats to feed, sexually mature and spawn, rendering them highly vulnerable to potential barriers like bridge aprons, culverts, flood gates and pumping stations.

The aims of this study were to describe and evaluate the impacts of a range of small-scale anthropogenic structures and associated environmental variables on the abundance and the diversity of sympatric fish communities along 16 lowland waterways across the Hawke's Bay region of New Zealand, so that migratory barriers can be identified and remediations can be implemented.

Up to six fine meshed fyke nets were set above (upstream) and below (downstream) various drainage structures along each waterway, and the abundance of all caught individuals were counted, taking into consideration any potential ethical and sampling issues. Generalised linear models were used to statistically analyse the impact of each structure on the abundance of fish, whilst Shannon Weiner, Pielou's evenness and species richness indices assessed the impact of each structure on the diversity of fish communities. Associated environmental measures were also recorded and were statistically analysed using mixed effect models to determine their effect on fish abundances.

The results of this study found that a bridge apron, a culvert and a 'Fish-Friendly Flood Gate' (FFFG) did not act as barriers to upstream migrating inanga. The structures also did not restrict the upstream passage of various other migrating and resident fish as a higher total abundance of all species was found upstream, and biodiversity indices were similar above and below the structures. In contrast, this study found that flood gates and pumping stations do act as full migratory barriers to inanga and other native fish, including the catadromous grey and yellow-eye mullet. Although the total abundance of all species was higher above these structures, the biodiversity was lower, as fish communities mainly comprised of the invasive resident mosquitofish, or the environmentally tolerant New Zealand freshwater shrimp.

The abundance of inanga and the total abundance of all species were also impacted by varying environmental factors, although the differences varied according to the type of structure present and whether the structure acted as a migratory barrier. Inanga abundances significantly increased with increasing water temperature, though significantly decreased with increasing dissolved oxygen (DO) saturation, whilst increasing specific conductivity (SPC) did not have an effect on inanga abundances. The total abundance of all species significantly increased with increasing temperature and SPC, whilst abundances were found to decrease with increasing DO saturations.

The findings of this study can be used to prioritise the mitigation or retrofitting of barrier structures on the site-specific level. They can also be applied on local and national levels so that comparable small-scale anthropogenic structures can be remediated to allow unrestricted upstream fish passage, with the aim of increasing native fish abundances and biodiversities, improved upstream habitat quality and the maintenance of healthy aquatic ecosystems.

Acknowledgements

Contents

List of Figures

Chapter 5 – Discussion

Chapter 7 – Appendices

List of Tables

Chapter 2 – Studied Site Descriptions

Chapter 3 – Methodology

Chapter 4 – Results

Chapter 7 – Appendices

Glossary

Term	Definition
Amphidromy	A form of diadromy involving the upstream drifting of juvenile fish from marine to freshwater environments to sexually mature and spawn. Newly hatched larvae drift downstream where they spend 3-6 months at sea.
Anadromy	A form of diadromy involving the upstream migrations of adult fish from marine to freshwater environments.
Anthropogenic	Or 'man-made'; resulting from the influence of human activity.
Brackish	Somewhat salty waters, which are neither considered freshwater, nor marine water, as in river estuaries.
Bycatch	Unwanted aquatic creatures caught within fishing traps or nets.
Catadromy	A form of diadromy involving the migrations of adult fish from freshwaters to marine environments.
Dendritic	A river system with many tributaries and sub-tributaries which are not straight in nature; the system resembles the branching pattern of tree roots.
Diadromy	A life cycle involving the migrations of species between marine and freshwater environments.
Ecosystem	A geographical area where a group of species live and interact with one another.
Ecotone	A boundary, or a transitional area between two ecosystems, often associated with a high species richness.
Endemic	A native species which is found only within a specific region or country.
Hypoxic	A waterway with low dissolved oxygen saturations.
Inter- and Intra-Specific Competition	**Inter** – competition between individuals of different species. **Intra** – competition between individuals of the same species.
Invasive/ Exotic Species	A non-native species, often introduced, which causes ecological and/or economic harm to an environment and the species within it.
Irrigation	In agriculture, the process of watering land to assist in crop production.

Juvenile	In aquatic zoology, a small-bodied individual with weaker swimming abilities which has not yet sexually matured into its adult form.
Keystone Species	A species which is considered essential for the healthy maintenance of an entire ecosystem. Without the keystone species, the ecosystem may collapse.
Lentic	An ecosystem with a body of still, standing water; varying in size from small pools to large lakes.
Lotic	An ecosystem with moving or flowing water, including streams and rivers.
Macrophyte	Large, aquatic plants; can be submerged, emergent, or floating.
Migratory Species	A species which moves from one habitat to another in order to complete its life cycle.
Mitigation	The process of reducing the severity or the negative effect by rectifying or resolving an issue.
Native Species	A species which has become a part of an ecosystem naturally.
Non-Tidal	A waterway which is not impacted by the natural fluctuations of the tide.
Outlier	In statistics, an observation which significantly differs from the general trend of the data.
Overdispersion	In statistics, when a greater variation is observed within a dataset than would be typically expected.
Potadromy	A life cycle involving the migrations of species along freshwater environments only.
Quantitative Data	In statistics, data which is in the form of numerical counts.
Resident Species	A species which permanently inhabits an ecosystem.
Riparian Vegetation	The zone between terrestrial and aquatic habitats which often holds a range of plant communities.
Sympatry	Species, communities, or populations which coexist within the same geographical area at the same time.
Tidal	A waterway which is directly impacted by the natural fluctuations of the tide.
Zero-inflation	In statistics, when the number of observed zeros within a dataset is higher than the number of predicted zeros.

Chapter 1

Introduction

1.1. The Importance of Free Movement for Migratory Fish

Worldwide there are over 10,000 described freshwater fish species, making it the most diverse, yet most threatened vertebrate group (Collen *et al*., 2014). He *et al*. (2019) documented that freshwater fish populations have declined by 81% within the past 50 years with 76% of these displaying various forms of migration within their life cycle. These declines are stated within the literature to be a cause of habitat loss, over-exploitation, climate change and human induced habitat fragmentation (Duncan and Lockwood, 2001; Vörösmarty *et al*., 2010). The free movement of migratory fish into upstream freshwater habitats plays a key role in the functioning of a healthy aquatic ecosystem (Kovach *et al*., 2015). As fish migrate from the sea, they transport and deposit essential marine nutrients through waste products, mortality or gamete deposition as they arrive in their upstream spawning habitats (Kovach *et al*., 2015). This is particularly evident in the Pacific salmonids as documented by Denton *et al*. (2010) who found that fish invest 30-70% of their body mass in nutrient- and energy-rich eggs deposited within riparian vegetation along upstream riverbanks. Bentley *et al*. (2012) further explained how fish eggs have a relatively low survival rate as they act as a key food resource to large fish, invertebrate, mammal and bird species, supporting a wide range of aquatic and terrestrial food webs (Childress and McIntyre, 2015).

Fish that migrate between salt and freshwater ecosystems as part of their life cycles are considered diadromous, migrating in search of suitable habitat to feed, spawn, reproduce, shelter from predators and escape unfavourable environmental conditions (Meixler *et al*., 2009). The term diadromous is then divided into 3 sub-categories: anadromous involving large-bodied adult individuals migrating from marine environments into freshwaters to reproduce; catadromous where the adult individual migrates from freshwaters into marine habitats to reproduce (Lassalle *et al*., 2008) and amphidromous which involves small-bodied, weak swimming juveniles drifting from marine environments into freshwaters where they feed, sexually mature and eventually reproduce. Newly hatched larvae then drift downstream into saltwaters, usually for a period of a few months to feed and grow (Franklin and Gee, 2019). Other migrant groups with different life cycles to diadromous species may also be vulnerable to changes in their environment, like the potadromous group who migrate entirely along freshwater systems. As studies discussing the impacts that barriers have on fish have focused on anadromous salmonids, anadromy has become the 'default setting' when attempting to analyse and mitigate issues caused by barriers. The focus on large-bodied adult anadromous salmonids fails to consider the migratory behaviour, the swimming capabilities and the life histories of small-bodied species, leading to poor and biased conservation attempts (Birnie-Gauvin *et al*., 2019).

The natural dispersal of diadromous fish along a river network has also been described in the literature as having a significant influence on human settlement (Lynch *et al*., 2016). Recreational and commercial fisheries are often located close to upstream spawning grounds and have provided high-nutrient food and economic security for local people for hundreds of years (Nieminen *et al*., 2017). Therefore, it must be emphasised that anthropogenic barrier construction may affect more

than 'just' a single species of fish and given the importance of migratory fish populations to the ecosystem as a whole, it is critical to understand the exact impact of specific barriers so that remediation efforts can be prioritised, and populations can be saved.

1.2. Habitat Fragmentation

1.2.1. The Causes and Effects of Habitat Fragmentation

Habitat fragmentation has been meticulously studied for decades, though much of the literature focuses primarily on fragmentation within a terrestrial setting (Crooks *et al*., 2017; Merckx *et al*., 2018). Habitat fragmentation occurs as large continuous habitats are broken into smaller, more isolated sub-habitats, or fragments, usually as a direct result of habitat loss or the loss of connectivity through natural or anthropogenic barriers (Crooks *et al*., 2017). Habitats dividing into smaller patches have been widely documented as having adverse effects on widespread vertebrate species across all ecosystem types, with maximum effects being shown on species with high mobility and dispersal to complete their life cycles (Collevatti *et al*., 2020). Recent studies by Levy *et al*. (2019) and Collevatti *et al*. (2020) describe how fragmentation can directly alter, and in some cases prevent, species dispersal whilst providing evidence of the detrimental effects fragmentation often has on population connectivity and genetic diversity within an ecosystem. Crooks *et al*. (2017) continues these ideas by explaining how isolated populations often result in significant changes to community and ecosystem composition, leading to a decline in species interactions and in some cases, species richness.

As high-quality ecosystems fragment into smaller patches, poor quality edge habitats become more numerous (Dias *et al*., 2013). Hermoso *et al*. (2011) and more recently Fuller *et al*. (2015) documented the effects that poor-quality fragmented habitats can have on specific species whilst also affecting the community as a whole. They described how habitat degradation promotes an increase in the distribution and local abundance of invasive or exotic species, as well as causing the loss of suitable breeding, spawning and refuge sites for native fish. This inevitably leads to increased exposure and predation risks for native species by invasives (Didham *et al*., 2007), highlighting the fact that habitat fragmentation, in any setting, comes with numerous direct and indirect negative impacts.

1.2.2. Fragmentation of Freshwater Systems

It is widely accepted in the literature that despite constituting only ~0.5% of the world's freshwater ecosystems, river systems are amongst the most threatened habitat types, suffering from numerous human induced impacts, including pollution and over exploitation (Wohl, 2005; Poff, 2014; King *et al*., 2020). However, an increasing number of recent studies have highlighted the impact that habitat fragmentation has on the health of a river network. King *et al*. (2020) described that river fragmentation can degrade a habitat, modify the water's flow regimes and affect a waterway's physio-chemical conditions, which subsequently can lead to a change in a river's aquatic biota. Species inhabiting a linear ecosystem, like those in a river or a stream, are at a higher risk of succumbing to the pressures of habitat fragmentation and reduced river connectivity compared with those inhabiting terrestrial environments (Cumming, 2004). Wohl (2017) suggested this was due to the continuous and dendritic nature of a river system and how its connectivity can be disrupted longitudinally, laterally or vertically. As well as this, Dias *et al*. (2013) and Fuller *et al*. (2015)

documented that poor-quality edge habitats in river networks can extend to substantial distances up and downstream, claiming that in the most extreme cases, these edge habitats can continue for hundreds of kilometres.

Ensuring that river networks remain longitudinally and laterally connected is essential in order to maintain a healthy ecosystem, as aquatic species with high mobility and lifetime dispersal require these connected habitats in order to survive, reproduce and maintain high population numbers (Bellucci *et al*., 2020). However, the loss of longitudinal connectivity in river systems is becoming a major problem worldwide (Segurado *et al*., 2015). Habitat fragmentation along a river is often caused by a single barrier, which has the potential to immediately isolate habitats from one another (Branco *et al*., 2012). Geist and Hawkins (2016) stated that the impacts of a single local barrier along a river can scale up to affect whole freshwater catchments, coastlines and even regional bays and seas, emphasising the destructive impact that habitat fragmentation can have on aquatic ecosystems. Numerous studies, including those by Abell *et al*. (2008) and Vörösmarty *et al*. (2010) named urbanisation, industrialisation and irrigation schemes as the major causes of connectivity loss in aquatic environments and that the degree of loss will only worsen as the human population increases.

1.3. Barriers to Fish Migration

1.3.1. Natural Vs Anthropogenic Barriers

Silva *et al*. (2018) defined a river barrier as a blockage which impedes the movement of organisms between up and downstream habitats, whilst altering a waterway's environmental characteristics. The 'River Continuum Concept' (RCC) (Vannote *et al.,* 1980) explains that a longitudinal river network consists of multiple linked ecosystems where the biota and physico-chemical processes downstream are directly affected by those upstream. Therefore, if a barrier alters the rates of carbon flow, sediment cycling or invertebrate drift upstream (as examples), similar changes will also occur downstream of the barrier (Richey and Swanson*,* 1989). A barrier may be natural, like waterfalls or cascades, physical, often anthropogenic, like dams or culverts or physico-chemical, inducing hydraulic, chemical or thermal changes (Araújo *et al*., 2018). It has been well documented in the literature that barriers cause habitat degradation and fragmentation (Van Puijenbroek *et al*., 2019), though it has also been accepted that successful migration across a barrier is highly species-specific, depending largely on the species' morphology, swimming and jumping capabilities and the barrier's physical structure (Rodgers *et al*., 2017).

Fuller *et al*. (2015) documented that waterfalls and cascades are some of the most numerous natural barriers whilst riparian vegetation falling into a waterway can also act as a significant barrier for some aquatic species. A study by Van Puijenbroek *et al*. (2019) shows how fragmentation from an initial physical barrier upstream can cause further physico-chemical barriers downstream by altering the flow regime, increasing water temperature, lowering dissolved oxygen (DO) levels and increasing the volume of suspended matter within the waterway. However, not all aquatic species are negatively affected by natural barriers; some are able to tolerate these changes, whilst others like the invasive mosquitofish *(Gambusia affinis)* can actually exploit the new conditions to their advantage (Fuller *et al*., 2015).

In contrast to natural barriers, the role anthropogenic barriers play on the connectivity of a river network has been widely documented, so much so that habitat fragmentation caused by manmade artificial structures is now recognised to have a direct role in the reduction of numerous freshwater

and migratory fish species (Meixler *et al.,* 2009; Januchowski-Hartley *et al.,* 2013). Unlike naturally occurring fragmentation, anthropogenic stressors on a waterway have advanced so rapidly that many aquatic species have been unable to evolve in order to cope with the changing environment (Fuller *et al.*, 2015). Richter *et al.* (1997) and Auster (1998) documented that >40,000 large dams worldwide, constructed within just 40 years, have divided important waterways into "punctuated staircases of slack water reservoirs", transforming diverse ecosystems holding pools, riffles, shallow banks and shoals into simplistic and poor-quality habitat. However, it must be emphasised that smaller waterways are often just as affected by anthropogenic barriers as larger systems; in fact, more recent studies by Abell *et al.* (2008) and Vörösmarty *et al.* (2010) concluded that ~65% of inland freshwater systems, of all sizes, are considered moderately to highly threatened by human activity. These anthropogenic stressors often present as drainage structures, road construction and flood protection schemes (Fischer and Lindenmayer, 2007). The impacts of these stressors on river habitats continues to increase as existing barriers develop and new structures are erected (Dias *et al.*, 2013). The numerous studies discussed so far in this review emphasise that the impacts of anthropogenic barriers are not merely theoretical but do in fact present real-life challenges which migrating fish species seem unable to easily overcome.

1.3.2. Large-Scale Vs Small-Scale Anthropogenic Barriers

There has been a considerable amount of literature published within the past 50 years which have explored the various impacts anthropogenic barriers have on river systems, though the vast majority have focused only on the effects of large-scale barriers, like dams (Cumming, 2004; Limburg and Waldman, 2009; Rincón *et al.*, 2017; Krieg and Zenker, 2020). As previously stated, large hydroelectric dam construction advanced considerably during the second half of the 20th century (Liermann, *et al.*, 2012), so much so that a study by Poff and Schmidt (2016) estimated that globally, there are ~58,000 large dams reaching over 15 metres in height, obstructing almost half of the world's freshwater river systems. Dams are primarily constructed to store water supplies which are used for irrigation and hydropower production, an increasingly popular method of obtaining renewable 'green' energy (Poff and Schmidt, 2016). It is commonly stated in the literature that dams cause significant and lasting damage to the river network they are constructed within by fragmenting the waterway in such an impactful way, that any form of connectivity between the newly created patches can be completely impossible.

Across various developed countries, small-scale barriers like culverts, flood gates and pumping stations exceed large-scale barriers by up to a thousand times (Liermann *et al.*, 2012). A study by Januchowski-Hartley *et al.* (2013) showed that there are 38 times more culverts as there are dams in the North American Great Lake basin signifying the vast quantity of small-scale barriers present along global freshwater systems. Culverts, the most common small-scale barrier, are constructed beneath road crossings within low-order streams with the aim of retaining some degree of water connectivity between habitats (Anderson *et al.*, 2014), whilst flood gates and pumping stations are erected for irrigation and flood control purposes (Liermann *et al.*, 2012). Most older small-scale barrier studies focused almost exclusively on large-bodied salmonids with strong swimming and jumping abilities (Park *et al.*, 2008; Birnie-Gauvin *et al.*, 2019), though an increasing number of newer fish passage schemes have considered the impacts on all migrating fish, as well as numerous amphibian, invertebrate and macrophyte plant species (Bradley *et al.,* 2012).

The physical impacts of large-scale dams on migrating fish species is well-documented, though a considerable proportion of these studies focus on large-bodied, economically important salmonids, mainly salmon and trout species (Mueller *et al.*, 2011). It is widely accepted that dams act as either full or partial physical barriers, disrupting the free movement of organisms and preventing access to suitable upstream habitat required to feed, reproduce and shelter (Liermann *et al.*, 2012; Radinger and Wolter, 2014). A recent study by Krieg and Zenker (2020) revealed that barriers across Europe have even been constructed within waterways with the primary aim of reducing migration rates and population numbers of invasive crayfish species, though evidence suggest these barriers do not differentiate between species, and local natives have also suffered population declines.

A considerable number of North American and European studies have discussed how Atlantic salmon *(Salmo salar)* and Chinook salmon *(Oncorhynchus tshawytscha)* populations have declined or disappeared in water systems that have been affected by human-induced habitat fragmentation (Schick and Lindley, 2007; Williams *et al.*, 2012; Nieminem *et al.*, 2017; Van Puijenbroek *et al.*, 2019). Each study discusses how large-scale barriers have led to habitat dysconnectivity, migration prevention and an increase in demographic isolation between populations. The Penobscot River in Maine, USA has been subject to numerous recent fish passage schemes as historic studies have shown that populations of returning Atlantic salmon stood at 75'000-100'000 per year, however by 2014, only 248 individuals returned to the river to spawn (Stevens *et al.,* 2019). Drainage barriers are abundant along the Penobscot River, though once 2 large dams were removed and numerous fish passage improvements were implemented, returning salmon populations increased to 1426 by July 2020 (Department of Marine Resources, 2020). It must be emphasised however that even though barrier construction was a major factor in the historic decline of Atlantic salmon, continuing population declines may be due to various other natural or anthropogenic factors unrelated to barriers.

A few North American studies have focused on the indirect impacts that large-scale dams have on migrating salmonids. McIntosh *et al.* (2010) explained that as migration routes are physically blocked by anthropogenic barriers, an unnatural concentration of fish accumulate around the barrier's edge. A study by Silva *et al.* (2018) discussed how this can lead to an increased predation risk for salmonid species by bald eagles *(Haliaeetus leucocephalus)*, bobcats *(Lynx rufus)*, river otters *(Lontra canadensis)* and larger-bodied fish species like the sea-run brown trout *(Salmo trutta).* Decomposing fish carcasses release a high quantity of marine nutrients and energy into freshwater ecosystems, crucial in maintaining a healthy chemical balance within the waterway (Willson and Halupka, 1995). The unnatural accumulation of fish downstream of a barrier can prevent these key nutrients from being released into the upstream environment, leading to increased unsuitable and degraded habitat (Silva *et al.*, 2018). A high abundance of fish in a relatively small area also increases the rate of inter- and intra-specific competition for ecological resources like food and habitat (McIntosh *et al.,* 2010), as well as becoming a hotspot for disease transmission (Brevé *et al.*, 2014), emphasising the indirect pressures migrating salmonids can also face when a habitat is anthropogenically fragmented.

Flood gates, pumping stations and lesser known small-scale barriers like weirs and fords are rarely acknowledged in the literature as having an impact on fish migrations. Within the past ~10 years however, culverts have become a focal point when discussing small-scale barriers in freshwater systems. Januchowski-Hartley *et al.* (2013) and more recently King *et al.* (2020) discussed that a culvert is often regarded as a partial barrier to migrating salmonids, so long as the structure's outlet

is not raised too far above the waterbed that individuals cannot swim or leap through the opening. It has also been reported that culvert slope, length and diameter can influence how 'passable' a barrier is (Anderson *et al*., 2014). It is evident in the literature that a fast flow velocity through a culvert acts as the most significant barrier to fish migration (Doehring *et al*., 2011a; Louca *et al*., 2014; Goerig *et al*., 2016). Faster flows create more turbulent environments which can become impassable if the velocity exceeds a fish's swimming capabilities (Doehring *et al*., 2011a). Even if the opening of the culvert is relatively accessible, fast flowing water can fatigue or injure migrating fish, lowering their motivation to seek out favourable spawning habitat upstream (Haro *et al*., 2004; Goerig *et al*., 2016). A study by Louca *et al*. (2014) suggested that some migrating fish delay or even avoid entering dark environments, like those found within longer-length culverts, further reducing successful passage rates.

A study by Kemp and O'Hanley (2010) described how the assessment criteria used to determine the passability of a barrier is based on large-bodied adult salmonids, ignoring other life stages and smaller species who tend to have weaker swimming abilities. Ovidio *et al*. (2007) found that 100% of tracked adult brown trout *(Salmo trutta)* were able to pass through a culvert in one area of the River Meuse in Belgium, yet in another area, only 33% of trout passed. This study provides evidence that a) different barriers cause different impacts on fish migration, and b) some barriers even restrict the movement of large-bodied fish. Therefore, it is important to understand the impact that these barriers may have on higher risk, smaller-bodied fish with weaker swimming and leaping capabilities.

1.3.2.2. Environmental Impacts of Barriers

Large-scale dams affect the surrounding environment from the initial stages of construction. Recent studies have described the impact that large-scale barriers can have on the environment; dams often create artificial upstream reservoirs which distorts the natural flow of the waterway and increases sediment transport downstream (Poff and Schmidt, 2016; Rincón *et al*., 2017). Vast areas of quality freshwater spawning and rearing sites are destroyed and replaced with simplistic, poor-quality gravel beds, whilst old coniferous riparian vegetation is removed and often replaced with young exotic plants or grazed banks (Sheer and Steel, 2006; Lenders *et al*., 2016). A number of studies (Cedarholm *et al*., 1999; Johnson *et al*., 2008; Liermann *et al*., 2012; Piccolo *et al*., 2012) documented that well oxygenated lotic habitats with little macrophyte cover are replaced by weedier lentic habitats with lower DO levels when river networks are anthropogenically fragmented. The literature also provides evidence that barriers often alter an ecosystem's water temperature and flow of nutrients, contributing towards the reduction of critical habitat required by migrating fish (Mueller *et al*., 2011; Gibson *et al*., 2017). Dauble *et al*. (2003) studied the impacts of habitat loss due to dam construction along two rivers in North America: the Columbia and the Snake, finding that only 13% and 58% of suitable habitat, respectively, for migrating species remained. A later study by Sheer and Steel (2006) reported similar findings along the same rivers, though added that ~70% of the remaining quality riparian habitat was located upstream of a large dam, rendering this habitat inaccessible for migrating fish species. These studies provide evidence that large-scale barriers degrade downstream habitat whilst also preventing access to suitable upstream habitat for migrating fish.

Studies showing the environmental impacts of small-scale anthropogenic barriers often mirror those exhibited by large-scale barrier studies, explaining how small-scale structures were also found to impede access to suitable upstream habitat (Ovidio *et al*., 2007; McLaughlin *et al*., 2013; Goerig *et al*., 2016). Van Puijenbroek *et al*. (2019) and King *et al*. (2020) documented that when culverts act as

either full or partial barriers, the water levels within the river network become low and channel incisions, levees or dikes can form which may act as further natural barriers to migrating fish. Therefore, it is essential to understand the passability of individual small-scale barriers in order to fully understand the impacts they have on migrating and resident fish species.

1.3.2.3. Genetic Impacts of Barriers

Studies discussing the genetic impact of natural and anthropogenic barriers on migrating fish are becoming more numerous in the literature. As habitats fragment, populations up and downstream of a barrier become demographically isolated (Roberts *et al*., 2013; Belliard *et al*., 2018). Studies by Haidvogl *et al*. (2015) and Rivers-Moore *et al*. (2016) found that this often leads to a reduction in genetic diversity within and between populations, resulting in an increase in vulnerability to environmental stochastic events and even possible local extinctions. Ferguson *et al.* (2019) further explained this by showing that demographic isolation can also lead to syntopy, a form of sympatry involving unrelated species. Temporal and behavioural differences in spawning may also occur which can subsequently result in genetic and reproductive isolation. Wofford *et al*. (2005) identified that barriers along the Camp Creek River in Oregon, USA affected the genetic structure of cutthroat trout *(Oncorhynchus clarkia)* by accelerating genetic differentiation and increasing population inbreeding and genetic drift (Schinegger *et al*., 2012).

1.4. New Zealand's Freshwater Species

1.4.1. Migratory Fish Vs Anthropogenic Barriers

A large proportion of fish species across all families inhabiting subtropical or temperate islands are migratory. New Zealand is no different as 18/35 indigenous freshwater fish species are classed as diadromous, with amphidromy being the most prevalent life history type (Franklin and Gee, 2019). In contrast, thoroughly studied diadromous European and North American species are considered anadromous and have significantly different passage requirements to the small and cryptic species that migrate through New Zealand's river networks (Franklin and Bartels, 2012). Franklin and Gee (2019) found that over the past few decades, New Zealand's waterways have faced increasing pressures from anthropogenic barriers resulting in widespread habitat degradation, flow regime alteration and ecosystem fragmentation. However, it wasn't until the 1990s that the effects of these barriers on native New Zealand fish were given some attention (McIntosh *et al*., 2010) and even today, only ~30% of large- and small-scale barriers across the country have been assessed for fish passage (Franklin and Gee, 2019).

Studies evaluating the impact of drainage barrier construction along New Zealand's waterways have focused mainly on large-bodied species like the pouched lamprey *(Geotria australis)* or the longfin eel *(Anguilla dieffenbachii)* and in some cases have included the small-bodied, amphidromous torrentfish *(Cheimarrichthys fosteri).* Jellyman and Harding (2012) revealed that other than the species named above, very few studies have focused on other native New Zealand fish, their passage requirements and the impacts migration barriers may have on their movement upstream. Jellyman and Harding (2012) themselves investigated the difference in species richness up and downstream of large dams along the Waikato River, New Zealand. They reported that a lower species richness was found upstream of the barrier which consisted mainly of exotic invasives whilst 10 diadromous species were captured in the downstream habitat. Though this study provides evidence that large-

scale barriers can have a detrimental effect on fish abundance in New Zealand, it fails to determine the impact of small-scale barriers on species biodiversity whilst also failing to consider which specific species were most affected.

1.4.2. The 'common galaxias', or inanga *(Galaxias maculatus)*

1.4.2.1. Amphidromy as a Life Cycle
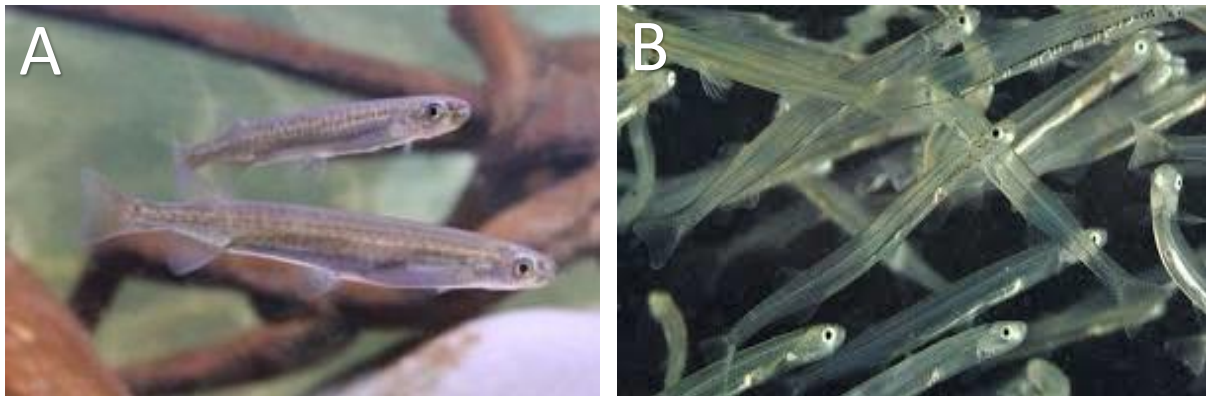
Figure 1.1. **A:** Adult 'common galaxias', or inanga *(Galaxias maculatus)* (HBRC, 2020). **B:** Juvenile inanga, the most common of New Zealand's whitebait species (DOC, 2020).

Widely distributed across the Southern Hemisphere, the Galaxiidae fish family are commonly found migrating through many of New Zealand's, Australia's, South America's and the Oceanic Island's lowland waterways (Stevens *et al*., 2016). Despite this, they remain one of the world's most endangered fish families where in New Zealand alone, >75% of the population is predicted to be threatened (McRae *et al*., 2016). The Galaxiidae family include New Zealand's 5 whitebait species, the most prevalent being the 'common galaxias' or inanga *(Galaxias maculatus)* (Fig.1.1) whose conservation status is classed as 'declining' (Dunn *et al*., 2018). It has been well documented in the literature that the amphidromous life cycle of the inanga begins as newly hatched larvae drift downstream from freshwater into saltwater environments where they spend a 3-6 month period out at sea, feeding and growing into juvenile 'whitebait' before drifting back upstream into freshwater habitats (Poulin *et al.,* 2012; McRae *et al*., 2018). Here, they feed and sexually mature into adult inanga, eventually spawning in riparian vegetation along streambanks (McRae *et al*., 2016). The inanga's life cycle does however depend largely on the ability to freely move up and downstream into suitable feeding and spawning grounds, which anthropogenic barriers may prevent. Inanga, along with the majority of New Zealand's other 18 indigenous diadromous freshwater fish, are small-bodied and weak swimmers with no ability to climb or leap across vertical barriers (Stevens *et al*., 2016). This renders them highly vulnerable to small-scale barrier implementation, an issue which large-bodied salmonids do not face. Therefore, it is essential to collate information regarding the passability of small-scale barriers for amphidromous species like inanga along New Zealand's waterways, rather than attempting to use previous data on large-bodied salmonids in the Northern Hemisphere.

1.4.2.2. The Importance of inanga

The annual migration of juvenile inanga, or whitebait is of significant recreational, commercial and cultural importance in New Zealand. Their recognition as a keystone species of many of the country's lowland freshwater ecosystems also highlights the inanga's ecological and environmental

significance (Lassalle *et al.*, 2008). Preying upon small insects, crustaceans and molluscs in the open water, inanga act as biological controls by removing pest species from their ecosystem whilst themselves acting as suitable prey to a range of large-bodied species. Rainbow trout (*Oncorhynchus mykiss),* the declining endemic New Zealand longfin eel *(Anguilla dieffenbachia)* and numerous wading birds are amongst those whose diet relies largely on inanga (Lassalle *et al.*, 2008). Migrating inanga transport large quantities of nitrogen and phosphorus in their tissues across long distances, which are released in dissolved forms through excretions and body decomposition (McIntyre *et al.,* 2007). Subsequently, these nutrient-rich solutions become readily available across multiple ecosystems for primary consumers like microorganisms and plants to absorb (McIntyre *et al.,* 2007), and so it is understood that the migration of inanga benefits species across the food web.

During periods of pre-European colonisation, indigenous Māori tribes exploited New Zealand's lowland waterways fishing for juvenile and adult inanga, marking the beginning of the large commercial whitebait fisheries which span the country today (Haggerty, 2007). As the centuries passed, access to waterways became easier and fishing technologies became more enhanced, allowing the recreational whitebait fishing market to expand drastically (McDowall, 1996). Numerous surveys have shown that from August – November each year, the typical 'whitebait season', the majority of those engaging in any activity on a river were whitebait fishers, whether commercial or recreational (Haggerty, 2007). In recent years, whitebait catch prices have reached up to NZ$100/kg (Haggerty, 2007), with inanga accounting for between 70-100% of New Zealand's average whitebait catch, and >95% of catches along the country's largest waterways (Yungnickel, 2017). Despite these findings, declines in the fisheries have been noted for several decades and are believed to be primarily due to overfishing, habitat destruction and drainage structure construction (Baker, 2006) which has resulted in the implementation of fishing regulations and restrictions in some areas of New Zealand (McDowall, 1984).

In 1840, the Treaty of Waitangi was signed between the British Crown and Māori chiefs, whereby it was promised that Māori traditions be protected and preserved (Orchard and Hickford, 2016). In Māori culture, whitebait is considered *taonga* (Ngāi Tahu Claims Settlement Act, 1998), a highly prized natural resource which should be respected, cared for and conserved (Haggerty, 2007). For tribes dating back to New Zealand prehistory, *taonga* food provided the basis of their annual harvest ceremonies, or *mana,* where they shared goods and fishing knowledge with neighbouring tribes (Haggerty, 2007). Inanga are also considered a *mahinga kai* species which describes them as a significant natural resource from an important aquatic environment. Presently, a range of regional and national governmental documents confirm in law that inanga are to be protected as a cultural priority (Orchard and Hickford, 2016). Therefore, the conservation of inanga is of high ecological, environmental, commercial and cultural importance.

1.5. Conclusion

This review has shown that the majority of the literature regarding human-induced waterway fragmentation focuses on the effects that anthropogenic barriers have on large-bodied salmonids with strong swimming and jumping abilities. A large proportion of these studies are based in the Northern Hemisphere along continental river networks where anadromy as a life history is the most prevalent. European and North American large-scale dams have been a focal point for research for decades with far less attention given to small-scale barriers. As urbanisation and road construction continues to rise, an increasing number of studies in the Southern Hemisphere have focused on the potential impacts that culverts may have on migrating fish, yet most continue to discuss the effects

on only large-bodied species. In New Zealand, the inanga *(Galaxias maculatus)* is an amphidromous small-bodied fish with a 'declining' conservation status, which in part, is due to the physical barriers that they face when attempting to migrate up and downstream. Despite their abundance along many of the country's lowland waterways, 70% of New Zealand's small-scale drainage structures have not yet been assessed for fish passability. The susceptibility of inanga to these potential barriers emphasises the need to examine the passability of each structure on an individual level so that sites can be prioritised, and effective remediation efforts can be implemented.

1.6. Aims and Objectives

The overall aim of this study is to describe and evaluate the impacts of a range of anthropogenic drainage structures and associated environmental variables on the abundance of inanga and the abundance and the diversity of sympatric fish communities.

The objectives of this study are therefore:

1. To sample and to describe the populations and the site variables,

2. To use these data to statistically test and evaluate the levels of impact of various drainage structures, and

3. To provide context to the results of the analyses and to subsequently make management recommendations.

The associated specific hypotheses tested in the study are set out in section 1.7.4.

1.7. Experimental Approach

1.7.1. Study Location and Present Drainage Structures

This study will examine a number of small-scale drainage structures, including pumping stations, flood gates and culverts to assess whether they act as potential barriers to migrating diadromous fish. Fyke netting will be set in upstream and downstream reaches of 16 lowland waterways across the Hawke's Bay region of eastern New Zealand. No drainage structures exist along five of the selected waterways and these sites will therefore be used as controls, whilst at least one drainage structure is present along each of the remaining 11 waterways. Sampling above (upstream) and below (downstream) of a drainage structure will test for differences in species richness and abundance, with particular attention given to adult and juvenile inanga *(Galaxias maculatus).*

Figure 1.2. The drainage structures used in this study are expected to look similar to those above: **A**: Chain Bridge Pumping Station, Lincolnshire, UK (Black Sluice Internal Drainage Board, 2016). **B**: Flood gate along the River Stiffkey, Norfolk, UK (Wright *et al.*, 2016). **C:** Pipe Culvert along Clugston Creek, Washington, US (INVW, 2019). **D:** Bridge Apron along the Cong Burn, UK (Wild Trout Trust, 2018).

Spanning a total area of 14,111 km² from Wairoa in the north to Woodville in the south, the Hawke's Bay region lies along the east coast of the North Island of New Zealand (Dowling, 1999). High levels of sunshine and annual precipitation levels of only 750-1000mm provide Hawke's Bay with a suitable climate for extensive agriculture, viticulture and horticulture, though practices like these can often lead to a degraded or transformed landscape (Dowling, 1999). Unreliable rainfall patterns during the winter months inflicts further environmental pressures onto Hawke's Bay, resulting in the implementation of numerous irrigation and flood defence structures within many of its waterways; in fact, as of 2020, 23 flood control schemes are running across the region (HBRC, 2020). The

amphidromous inanga migrate upstream along many of these anthropogenically fragmented river networks, including two of the region's largest: the Ngaruroro and the Tutaekuri. Hawke's Bay's climatic conditions, along with the fragmentation of its waterways, transformation of its landscape and presence of migrating inanga provides a suitable location for this study to be based.

The anthropogenic fragmentation occurring along many of Hawke's Bay's tidal and non-tidal waterways is mainly caused by the implementation of various small-scale drainage structures. The initial aim of these structures is to prevent flooding in lowland areas during periods of high rainfall, though a number are also constructed beneath road crossings to allow continued water flow between up and downstream habitats. In this study, each of the 11 sites where a structure is present will have at least one pumping station (Fig.1.2A), flood gate (Fig.1.2B) or culvert (Fig.1.2C). A single bridge apron (Fig.1.2D) is present along one of the sites which will also be assessed for fish passability.

Pumping stations, the largest sized structures of those examined in this study, regulate a river's water level and flow by pumping water downstream to maintain a lower river elevation in the upstream habitat (Bolland *et al*., 2018). When pumps are not operational, the flow of water along the upstream catchment decreases, often until stagnant, potentially impacting the ecosystem in a range of ways (Bolland *et al*., 2018). Decreased river complexity and alterations to its natural hydrology may lead to significant reductions in species diversity in the upstream habitat, shifting population abundances towards generalist or pest species (Briggs and Galarowicz, 2013). Although the passability of each pumping station in this study is not yet known, these structures generally do not have any inbuilt fish specific protection controls like exclusion grills or fish ramps (Bolland *et al*., 2018).

Used widely across the world, flood gates are constructed within tidal waterways which are prone to flooding. They provide significant flood protection for upstream agricultural land and human infrastructure by preventing the natural pulse of tidal water that frequently occurs in lowland coastal waterways from continuing upstream (Richardson and Taylor, 2002). During high or incoming tides, the flood gates close, inhibiting the passage of aquatic species into upstream habitats. As the tide recedes, the flood gates reopen, and water is drained from upland areas (Franklin and Hodges, 2015). Though the path for migrating species is restored when the gates reopen, fish must migrate against a strong water current and across the drainage structure in order to continue their migration upstream. A process which may not be possible for small-bodied species with weak swimming abilities like the inanga (Richardson and Taylor, 2002; Franklin and Gee, 2019), hence why it is crucial to assess the passability of each individual flood gate (Franklin and Gee, 2019).

As previously discussed, culverts are the most numerous small-scale barrier and are commonly used to allow continued passage of water beneath constructed roads and bridges (Briggs and Galarowicz, 2013). The shape and size of a culvert can vary according to the height of the roadway embankment, required hydraulic performance and quality of its surrounding habitat. Pipe culverts generally do not exceed 3 metres in diameter, whereas large-scale box culverts can span over 15 (Richardson and Taylor, 2002; Briggs and Galarowicz, 2013). Small sized pipe culverts are often installed within first-order streams beneath country or farmland roads, habitats which often act as spawning or resting grounds for native fish species (Doehring *et al*., 2011a). Culvert construction often focuses on optimising hydraulic conveyance, with little or no attention given to the biological requirements of migrating species which require passage in order to fulfil their life cycle (Franklin and Bartels, 2012). As culverts are constructed in a range of shapes and sizes and their effect on fish migrations may differ between each one, it is important to assess the passability of the structures at an individual level.

After extensive searching through the literature, this study is believed to be the first of its kind which prioritises the impacts of small-scale structures on small-bodied inanga in Hawke's Bay and New Zealand as a whole.

1.7.2. Environmental Analyses

As well as examining the abundance and the biodiversity of species upstream, midstream and downstream of various drainage structures, this study will also be analysing the effect, if any, of the water temperature, the dissolved oxygen saturation and the specific conductivity of each sub-site on the abundance of inanga and the total abundance of all species. Site-specific written habitat assessments will be included to influence local management solutions across the lowland waterways of Hawke's Bay.

1.7.3. Data Analysis and Remediations

This study is expected to yield count data containing a high number of zeros which must be considered whilst performing statistical analyses. Poisson, negative binomial and zero-inflated models will be amongst those used to test for significance in abundance and diversity data between upstream and downstream sub-sites. Similar models will be used to test for significance between each of the 16 sampled sites and mixed effect models will incorporate any environmental influences. All statistical analyses will be conducted to answer pre-specified hypotheses.

Once the degree at which each drainage structure acts as a barrier to upstream fish migrations has been assessed, remediation decisions can be made on how to allow passage across each barrier. This will enable sites with the most impactful barriers to be prioritised so that attempts to improve fish passability into upstream habitats can be implemented. As the drainage structures prementioned are commonly used in waterways around the globe, the data yielded from this study can be used as a basis for the remediation of waterways beyond Hawke's Bay, and indeed New Zealand. Though the distribution of inanga is far reaching and spans much of the Southern Hemisphere, this study's data showing the impact of barriers on inanga migration can also be used to determine the impacts on other small-bodied diadromous species. Therefore, this study has the potential to examine both the effects of site-specific drainage structures on upstream migrating inanga, as well as having a wider applicability to waterways and aquatic migrations around the globe.

1.7.4. Hypotheses

The aim of this study is to either accept or reject (when $p < 0.05$) the following hypotheses relating to the abundance, species diversity and environmental effects on inanga and all species combined across 16 lowland waterways in Hawke's Bay:

<u>Species Abundance:</u>

H_0: Inanga abundance will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'barrier status' or 'treatment'.

H_1: Inanga abundance will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'barrier status' or 'treatment'.

H_0: The total abundance of all species will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'barrier status' or 'treatment'.

H_1: The total abundance of all species will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'barrier status' or 'treatment'.

<u>Species Diversity and Similarity</u>:

H_0: Biodiversity, species richness and species evenness upstream (and midstream) will not differ to the biodiversity, species richness and species evenness downstream if fish passage is unrestricted along waterways with varying 'barrier status' or 'treatment'.

H_1: Biodiversity, species richness and species evenness upstream (and midstream) will differ to the biodiversity, species richness and species evenness downstream if fish passage is restricted along waterways with varying 'barrier status' or 'treatment'.

<u>The Effects of Environmental Factors on Species Abundance:</u>

H_0: Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the abundance of inanga along waterways with varying 'barrier status' or 'treatment'.

H_1: Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the abundance of inanga along waterways with varying 'barrier status' or 'treatment'.

H_0: Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the total abundance of all species along waterways with varying 'barrier status' or 'treatment'.

H_1: Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the total abundance of all species along waterways with varying 'barrier status' or 'treatment'.

Chapter 2
Studied Site Descriptions

The ecological impacts of anthropogenic drainage structures on the abundance and diversity of inanga and other fish species display common principles but are usually modified by site-specific habitat and environmental variables that influence local management solutions. The location, the drainage structure present, if any, and the main environmental features of the upstream, midstream, if applicable, and downstream sub-sites are described for the 16 tidal and non-tidal lowland waterways across Hawke's Bay identified and selected for assessment in this study, as shown in Fig.2.1. The Cawthron Institute's (2015) 'National Rapid Habitat Assessment Protocol Development for Streams and Rivers' habitat survey form was used as reference for the written descriptions of each sub-site.

2.1. Selected Study Sites

Whilst liaising with the Hawke's Bay Regional Council (HBRC) in New Zealand, sites which were either known or believed to be used by inanga were chosen with additional consideration given to their accessibility and security, i.e., the ability to leave netting within the waterway for prolonged periods of time without the risk of being tampered with or removed by members of the public. At least one drainage structure is present along 11 of the 16 sites and no drainage structure is present along the remaining five sites which were categorised as controls.

The drainage structures present are either bridge aprons, culverts, flood gates or pumping stations. The abundance and the diversity of species were assessed in upstream and downstream sub-sites along each of the 16 chosen waterways. However, two sites also had an additional midstream sub-site where a second potential barrier was observed and assessed, noting that in this study, the term 'midstream' refers to an intermediate sub-site which is located between the upstream and the downstream sub-sites, rather than referring to an area halfway across a water channel. Further detail and additional descriptions of terms used in this study are set out in section 3.2.2.

For sites with a present structure, the upstream sub-site lies above the structure, and the downstream below the structure. The majority of the sites used in this study are HBRC owned and therefore, when liaising with the council, no access permissions or permits were required for most sites. Some however are privately owned and did require external permissions, though easy access to landowner details through the council allowed for efficient phone calls to be made requesting access to private land. All requests were granted and data collection across all 16 sites could continue.
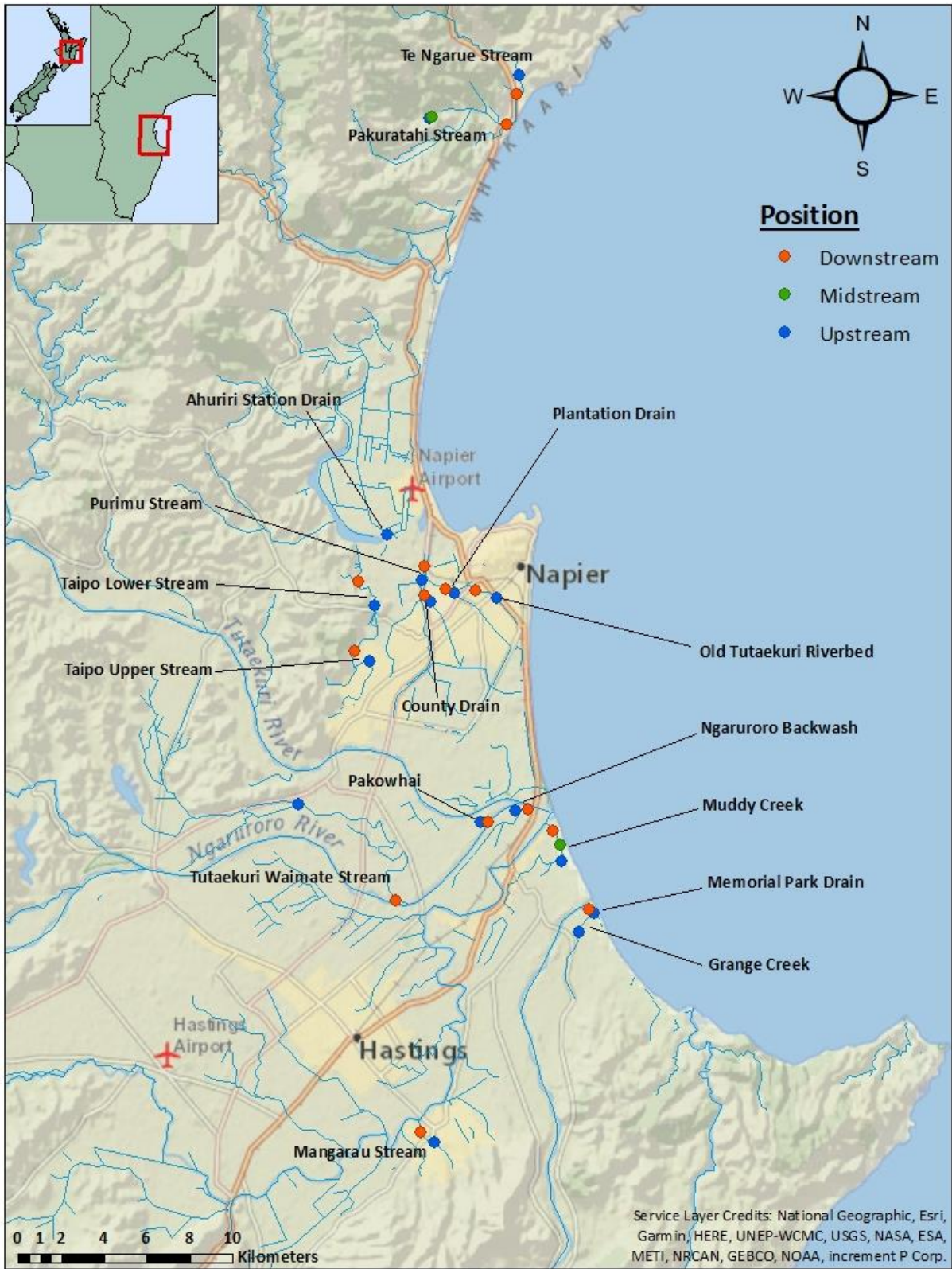
Figure 2.1. ArcMap showing the 16 studied lowland waterways across Hawke's Bay, New Zealand, in relation to the cities of Napier and Hastings. The legend shows each waterway's upstream, midstream, where applicable, and downstream sub-sites.

Each site has been assigned a 'short code' which will be used in future tables and habitat descriptions. Each site has also been categorised into 'barrier status' and 'treatment' (Table 2.1) which will act as two of the independent variables used in the analyses of the results in this study. The sampling design and definitions of each of the 'barrier status' and the 'treatment' categories, or "levels" are described in detail later in this study.

Table 2.1. Full names, short codes, barrier status and treatment type for the 16 lowland waterways sampled in this study. PS = Pumping Station, FFFG = Fish-Friendly Flood Gate. Refer to Methods section 3.2.2. for sampling design and definitions of the barrier status and treatment terminology.

Studied Site Full Name	Short Code	Barrier Status	Treatment
Upper Taipo Stream	taup	Control	Control
Lower Taipo Stream	talm	Control	Control
Grange Creek	grcr	Control	Control
Te Ngarue Stream	tnst	Control	Control
Pakuratahi Stream	past	Control	Control
Mangarau Stream	mgst	Non-Barrier	Bridge Apron
Ngaruroro Backwash	ngbw	Non-Barrier	Culvert
Tutaekuri Waimate Stream	twst	Barrier	Flood Gate
Pakowhai	paps	Barrier	Flood Gate
Memorial Park Drain	mpdr	Barrier	Flood Gate
Muddy Creek	mucr	Barrier	PS/ FFFG
County Drain	codr	Barrier	Pumping Station
Old Tutaekuri Riverbed	otri	Barrier	Pumping Station
Purimu Stream	pust	Barrier	Pumping Station
Ahuriri Station Drain	asdr	Barrier	Pumping Station
Plantation Drain	pldr	Barrier	Pumping Station

2.2. Habitat Site Descriptions

In order to contextualise the results of this study, it is important to firstly describe the features of each of the 16 sampled waterways so that site-specific remediations and local management solutions can be implemented. Brief written habitat descriptions of each of the upstream (US), the midstream (MS) and the downstream (DS) sub-sites are shown below. For each sub-site, an observational habitat survey form was completed, named the 'National Rapid Habitat Assessment Protocol Development for Streams and Rivers' and sourced from the Cawthron Institute (Clapcott, 2015). Summaries of the main findings of the forms and the main features of each sub-site are displayed in Table 2.2.

Further detailed written descriptions of each site, including the date of study, weather conditions, hydrographic tidal predictions, notes on the ease of access, detailed descriptions of the drainage structure present, GPS coordinates (DD) and in-depth descriptions of each sampled sub-site, as well as maps and photographs of the drainage structure and each sub-site is available in Appendix B.

Table 2.2. Summary habitat assessments of the upstream (US), the midstream (MS) and the downstream (DS) sub-sites for each of the 16 sampled waterways in this study. Physical features, hydraulic heterogeneity, vegetation and riparian measures assessed using the 'National Rapid Habitat Assessment Protocol Development for Streams and Rivers' form sourced from the Cawthron Institute (Clapcott, 2015). Refer to Methods Section 3.2.3 for descriptions on how the mean channel width (m) and depth (cm) measures were calculated.

Site Short Code and Position	Physical Features			Hydraulic Heterogeneity			Vegetation		Riparian			Local Land Use
	Mean Channel		Substrate Types	Hydraulic Component Variability	Flow Velocity	Macrophyte Density	Riparian		Width (m)	Shade	Erosion	
	Width (m)	Depth (cm)										
taup												
US	2.0	18.3	Fine sediment	Shallow pools	Slow	Low	Heavily mown/ mature native		2-5	Medium	High	Residential
DS	3.5	62.8	Fine sediment	Deep pools	Slow	Medium	Heavily mown/ long grasses		3-5	Low	Medium	Agricultural
talm												
US	7.2	81.3	Fine sediment/ cobbles	Deep pools	Slow	High	Heavily mown/ mature native		1-3	High	High	Recreational
DS	9.7	100.0	Fine sediment	Shallow pools	Slow	Medium	Heavily mown/ tussocks and sedges		2-4	Low	Medium	Agricultural
grcr												
US	5.9	45.1	Fine sediment	Shallow pools	Slow	Low	Heavily mown		2	Low	High	Agricultural
DS	8.1	48.1	Cobbles/ gravel and sand	Shallow pools/ riffles	Fast	Medium	Tussocks and sedges/ canopy trees		10-15	Low	Low	Recreational
tnst												
US	4.4	59.9	Cobbles/ gravel and sand	Deep pools/ riffles	Mixed	Low	Mature native		5	High-Medium	Medium	Agricultural
DS	9.6	131.8	Fine sediment	Deep pools	Slow	Low	Mature native		5-7	High	Medium	Agricultural
past												
US	1.9	43.2	Cobbles/ gravel and sand	Deep pools/ riffles	Mixed	Low	Long grasses/ mature native		5	High	Medium	Agricultural

MS	5.3	33.3	Fine sediment	Shallow- deep pools	Slow	High	Long grasses	7	Low	High	Agricultural
DS	18.7	-	Fine sediment	Deep pools	Fast	Low	Tussocks and sedges	5	Low	Medium	Agricultural
mgst											
US	2.3	32.1	Cobbles/ gravel and sand	Deep pools/ riffles	Mixed	Low	Mature native/ tussocks and sedges	4-5	High	Medium	Residential
DS	1.6	53.3	Cobbles/ gravel and sand	Deep pools/ riffles	Mixed	Medium	Long grasses	7-10	Medium	Medium	Residential
ngbw											
US	10.2	95.8	Fine sediment	Deep pools	Slow	Medium	Long grasses/ tussocks and sedges	7-10	Low	Medium	Agricultural
DS	25.4	68.5	Cobbles/ gravel and sand	Deep pools/ riffles	Mixed	Medium	Long grasses/ tussocks and sedges	7-10	Low	High	Agricultural
twst											
US	6.2	79.6	Cobbles/ gravel and sand	Deep pools	Slow	Medium	Long grasses	3-4	Low	Medium	Agricultural
DS	10.8	-	Cobbles/ gravel and sand	Deep pools	Fast	High	Long grasses/ mature native	5-7	Low	Medium	Agricultural
paps											
US	5.9	107.2	Fine sediment	Deep pools	Slow	High	Long grasses/ tussocks and sedges	4-7	Low	Low	Agricultural
DS	2.3	41.0	Cobbles/ gravel and sand	Shallow pools	Slow	Low	Long grasses	7	Low	High	Agricultural
mpdr											
US	3.0	68.2	Fine sediment	Deep pools	Slow	High	Heavily mown/ mature native	1-3	Medium	Medium	Residential
DS	8.1	48.1	Cobbles/ gravel and sand	Shallow pools/ riffles	Fast	Medium	Tussocks and sedges/ canopy trees	10-15	Low	Low	Recreational
mucr											
US	8.9	96.9	Fine sediment	Deep pools	Slow	Low	Long grasses/ tussocks and sedges	5	Low	Medium	Agricultural
MS	56.8	91.8	Fine sediment	Deep pools	Slow	Medium	Tussocks and sedges	3-7	Low	Medium	Recreational

DS	7.7	68.9	Fine sediment	Shallow-deep pools	Medium	Medium	Long grasses/ tussocks and sedges	3-7	Low	High	Recreational
codr											
US	7.1	100.5	Fine sediment	Deep pools	Slow	Low	Long grasses/ mature native	3	Medium	Medium	Industrial
DS	8.9	27.1	Fine sediment	Deep pools	Slow	Low	Heavily mown	2	Low	Low	Industrial
otri											
US	7.9	56.3	Fine sediment	Deep pools	Slow	Medium	Heavily mown	1	Medium	High	Recreational
DS	8.9	64.5	Fine sediment	Deep pools	Slow	High	Heavily mown	< 0.5	Medium	Medium	Recreational
pust											
US	16.6	85.4	Fine sediment	Deep pools	Slow	High	Heavily mown	5	Low	Low	Recreational
DS	9.7	100.0	Fine sediment/ cobbles	Shallow pools/ riffles	Mixed	Low	Long grasses/ tussocks and sedges	5-10	Low	Medium	Recreational
asdr											
US	8.3	123.8	Fine sediment/ cobbles	Deep pools	Slow	High	Long grasses/ mature native	5-7	High	High	Agricultural
DS	9.7	100.0	Fine sediment/ cobbles	Shallow pools/ riffles	Mixed	Low	Long grasses/ tussocks and sedges	5-10	Low	Medium	Recreational
pldr											
US	5.4	84.9	Fine sediment	Deep pools	Slow	Low	Heavily mown	4	Low	High	Industrial
DS	4.4	78.2	Fine sediment/ cobbles	Deep pools	Slow	High	Mature native/ tussocks and sedges	1-5	High-Medium	Medium	Industrial

2.2.01. Upper Taipo Stream

Located over 10km from the open ocean, the non-tidal upper Taipo stream branches south from the Ahuriri Estuary and continues upstream in a southerly direction for an additional 3km. There are no drainage structures present along the upper Taipo Stream. Mature, native trees with an intact undergrowth spans the upstream sub-site's left bank, providing habitat for aquatic species and casting a significant amount of shading across the stream, preventing excessive macrophyte growth. Overhanging vegetation has led to considerable bank slumping, which in some areas has resulted in landslips and the release of substrates into the water channel. Isolated patches of woody debris, floating macrophytes and partially submerged riparian vegetation span the length of the lentic downstream sub-site, though fluctuating water levels during periods of high rainfall have led to the undercutting of each bank.

2.2.02. Lower Taipo Stream

Situated almost 6km from the open ocean and continuing in a southerly direction for 6.5km, the tidal lower Taipo stream is also free of any drainage structures. Although mature, native trees along the upstream sub-site's right bank overhang and cast a significant amount of shading, the water channel remains densely choked with an excessive volume of exotic macrophytes like water buttercup *(Ranunculus* sp*.)* and curled pondweed *(Potamogeton crispus).* The slumping riparian vegetation has led to considerable erosion, resulting in undercut banks, the release of woody debris and the exposure of large root mats. Sedges, tussocks and brambles overhang from the downstream sub-site's banks, providing habitat for aquatic species and protection against erosion due to the fluctuations of the tide, though shading across the stream remains limited.

2.2.03. Grange Creek

Branching south from the downstream Memorial Park Drain sub-site, the upstream Grange Creek sub-site continues in a south westerly direction for an additional 4km. The downstream Memorial Park Drain sub-site is also used as the downstream sub-site for Grange Creek and there are no drainage structures present between the two tidal sub-sites. Heavily mown grass and the lack of sedges and tussocks along the upstream sub-site has exposed both of the left and the right banks to significant active erosion caused by the daily fluctuations of the tide, resulting in undercut banks and the release of cobbles, soil and woody debris into the water channel. Little shading is cast across the stream, though macrophyte growth remains minimal.

2.2.04. Te Ngarue Stream

Situated 1km from the open ocean, the tidal Te Ngarue stream runs in a northerly direction, east of the Tangoio forest. As there are no drainage structures present, the Te Ngarue continues freely upstream for at least 6km, though numerous additional tributaries may further increase the amount of available habitat. Mature, overhanging trees with diverse understories cast significant amounts of shading across the hydraulically diverse upstream sub-site. Active erosion caused by the slumping banks have released woody debris, cobbles and silt into the stream's deep, lentic pools and fast flowing, turbulent riffles and waterfalls. In contrast, the downstream sub-site has little hydraulic heterogeneity as a high volume of riparian vegetation has fallen into the stream's deep pools, slowing the water flow and reducing the amount of turbulence.

2.2.05. Pakuratahi Stream

Running in a westerly direction into the Tangoio forest, the tidal Pakuratahi stream lies 0.4km from the open ocean and continues above the upstream sub-site for a further 2km, though numerous tributaries branching from the stream may increase the amount of available habitat. There are no drainage structures present, though a small ford lies between the upstream and an additional midstream sub-site. Dense riparian grasses line each of the upstream sub-site's banks, overhanging across a fast flowing, turbulent riffle and numerous slow flowing, deep pools. A significant amount of shading is cast, whilst macrophyte growth is minimal. Long grasses and young shrubs span the midstream sub-site's left and right banks, though very little shading is cast across the stream, leading to the substantial growth of submerged and emergent macrophytes which choke the waterway. During a low tide, a slow flowing wetland habitat forms around the downstream sub-site's banks, though the water flow remains fast and turbulent across the remainder of the stream. A small amount of shading is cast from the overhanging, native trees which span the lower regions of the sub-site, though macrophyte growth remains minimal.

2.2.06. Mangarau Stream

Branching south from the Karamu stream, a tributary of the Clive River, the non-tidal Mangarau stream lies less than 20km from the open ocean and continues upstream into the Kohinurakau Hill Range for at least 7km. A bridge apron with a single drop ledge divides the upstream and downstream sub-sites. Large overhanging canopy trees and dense shrubs span the upstream sub-site's banks, casting a significant amount of shading across the stream and minimising the level of macrophyte growth. Curly pondweed *(Potamogeton crispus)* is found in abundance along non-shaded areas of the downstream sub-site, though remains sparse where shading is cast from overhanging riparian vegetation. The stream has high hydraulic heterogeneity as deep lentic pools and fast flowing, turbulent riffles are present along both sub-sites.

2.2.07. Ngaruroro Backwash

Branching in a south-westerly direction from the Waitangi Estuary, the tidal Ngaruroro Backwash site lies over 1km from the open ocean and runs parallel to the Ngaruroro River. Approximately 2.6km of available habitat lies above a medium-sized pipe culvert which divides the stream's upstream and downstream sub-sites. Dense long grasses and mature shrubs span the riparian buffers along the left and the right banks of the upstream sub-site, though little shading is cast across the stream, resulting in the excessive growth of floating and submerged macrophytes. Woody debris, root mats and eroded stream banks have formed slow flowing pools and fast flowing riffles across the width of the downstream sub-site. Shading remains sparse and macrophyte growth is excessive.

2.2.08. Tutaekuri Waimate Stream

Branching in a northerly direction from the Ngaruroro River, the non-tidal Tutaekuri Waimate stream lies over 7km from the open ocean and continues west for at least an additional 30km. Two medium-sized pipe culverts, each with an attached flood gate mechanism divides the stream's upstream and downstream sub-sites. Overhanging, exotic grasses and shrubs span the partially eroded left and right banks along the upstream sub-site, casting a small amount of shading and releasing large quantities of substrate into the stream. Despite the presence of mature, native trees which are scattered amongst long grasses along the downstream sub-site's left bank, little shading is cast, which has subsequently led to the extreme growth of submerged macrophytes which choke the waterway.

2.2.09. Pakowhai

Situated almost 3km from the open ocean, the tidal Pakowhai stream branches west from the Ngaruroro River, south of the Tutaekuri River and continues upstream for an additional 7.8km in a south westerly direction. A large pipe culvert, with an attached flood gate mechanism divides the upstream and the downstream sub-sites. Sedges, tussocks and heavily mown grass span each of the upstream sub-site's banks, casting very little shading across the stream, leading to the excessive growth of floating and submerged macrophytes which choke up significant areas within the waterway. Considerable amounts of woody debris, plant matter and large substrates line the stream bed along the downstream sub-site. Active erosion has led to the undercutting of each bank and the exposure of large root mats.

2.2.10. Memorial Park Drain

Situated 0.3km from the open ocean and branching south from the Tukituki River, the tidal Memorial Park Drain site continues upstream in a south easterly direction for approximately 1.9km. A medium-sized pipe culvert with an attached flood gate mechanism divides the upstream and downstream sub-sites. Overhanging, native trees are sparsely scattered amongst heavily mown grass which spans each of the upstream sub-site's banks, casting significant shading across some regions of the slow-flowing stream, though macrophyte growth remains high across the sub-site. Diverse riparian vegetation spans the downstream sub-site's left and right banks, providing some protection against tidal erosion and a large amount of habitat for aquatic species. The sub-site has high hydraulic heterogeneity as turbulent riffles and slow-flowing pools are formed amongst large boulders and cobbles as the tide recedes.

2.2.11. Muddy Creek

Located less than 1km from the open ocean, the tidal Muddy Creek branches south from the Clive River, before meandering in a westerly direction above the upstream sub-site for approximately 10km. A flood gate with an attached 'fish-friendly' mechanism divides the downstream and an additional midstream sub-site and a medium-sized pumping station divides the midstream and upstream sub-sites. Dense strips of overhanging shrubs, sedges and long grasses span each of the upstream sub-site's banks, though shading remains minimal, and a thick layer of green algae has accumulated around the pumping station. Native tussocks are abundant along the midstream sub-site's left and right banks, providing suitable spawning habitat for aquatic species and bankside protection against tidal erosion. Very little shading is cast across the stream, the water flow is slow and submerged macrophyte growth is moderate. Tidal fluctuations have caused significant erosion along some regions of the downstream sub-site, leading to undercut banks and the release of woody debris and fine sediment into the waterway. The lack of shading cast by either bank has led to the growth of macrophytes in isolated patches across the stream.

2.2.12. County Drain

Situated over 3km from the open ocean, the tidal County Drain site lies along the Tannery stream, which itself branches south from the Ahuriri Estuary. Approximately 6km of suitable habitat continuing in a south-easterly direction lies above a medium-sized pumping station which divides the County Drain's upstream and downstream sub-sites. Mature trees, shrubs and long grasses along the upstream sub-site's left and right banks encroach across the stream, casting a significant amount of shading and subsequently limiting the volume of macrophyte growth within the water channel. In comparison, heavily mown grass spans each of the downstream sub-site's left and right banks,

casting little shading and providing some suitable fish spawning habitat, though macrophyte growth within the stream remains low.

2.2.13. Old Tutaekuri Riverbed

Running in a south easterly direction over 5km from the open ocean, the tidal Old Tutaekuri Riverbed branches east from the Tannery stream, which itself branches from the Ahuriri Estuary. At least 1km of available habitat lies above a large pumping station which divides the stream's highly modified upstream and downstream sub-sites. Large slumping trees are scattered amongst heavily mown grass along each of the upstream sub-site's eroding banks, casting some shading across the width of the stream and releasing large quantities of woody debris, plant matter and substrate into the water channel. The downstream sub-site shows similar hydraulic and riparian properties as the upstream, though floating and submerged macrophyte growth is significantly higher, slowing the water flow and choking most regions of the stream.

2.2.14. Purimu Stream

Branching south from the Ahuriri Estuary, the tidal Purimu stream lies less than 3km from the open ocean and continues upstream for approximately 10km where it eventually joins the Tannery stream. A large pumping station divides the upstream and the downstream sub-sites. Heavily mown grass spans each of the upstream sub-site's highly modified banks, casting little shading and stimulating the excessive growth of a dense subsurface canopy of hornwort *(Ceratophyllum demersum),* an introduced, displacing macrophyte species. Tussocks and long grasses along each of the downstream sub-site's banks encroach into the stream, offering shelter for aquatic species and protection against bank erosion. During a low tide, fast flowing riffles form over small cobbles and boulders as macrophyte growth is minimal.

2.2.15. Ahuriri Station Drain

Located less than 1km from the open ocean, the upstream tidal Ahuriri Station Drain site branches north from the Ahuriri Estuary and continues in a north westerly direction for an additional 8.5km. The tidal downstream sub-site along the lower Taipo stream is also used as the downstream sub-site for the Ahuriri Station Drain site and lies 6km from the open ocean. A large pumping station divides the upstream and the downstream sub-sites. Mature, native trees, shrubs and long grasses cast a significant amount of shading across the upstream sub-site, though floating and submerged macrophyte growth remains high. Overhanging riparian vegetation has led to the active slumping of each bank and the release of large quantities of woody debris, plant matter and substrates into the water channel.

2.2.16. Plantation Drain

Running in a south easterly direction approximately 4km from the open ocean, the tidal Plantation Drain site lies along a tributary of the Tannery stream and continues upstream for an additional 3km. A medium-sized pumping station divides the upstream and downstream sub-sites. Heavily mown grass spans the upstream sub-site's highly degraded banks and significant active erosion has led to large landslips which have released soil, rock and plant matter into the stream. The sub-site's water quality is low as a number of drainage pipes leak unknown, oil-based liquids directly into the water channel. A diversity of native trees, flaxes, sedges and grasses span the downstream sub-site's left bank, casting shading across large areas of the stream, though submerged macrophyte growth is high in all non-shaded regions.

Chapter 3
Methodology

3.1. Justification of Approach

Several commonly used fish capture methods have successfully examined the abundance of aquatic species within lowland waterways, though none are without their own specific trade-offs. This signifies the importance of reviewing the pros and cons of each potential method and determining which is best suited for this particular study. Fyke netting, gill netting and electro-fishing are amongst the most widely used fish capture techniques, though the effect on fish mortality and habitat destruction differs between each approach.

Fyke netting is a passive sampling technique because it relies solely on fish willingly approaching and entering the nets with no additional force or bait required. Bycatch and fish mortality is minimal when exclusion grills are attached, and any unwanted catch can be released alive (Portt *et al.,* 2006). Small mesh sizes also reduce the risk of injury as caught fish are less likely to become trapped or entangled whilst attempting to escape. In comparison, bycatch and fish mortality is high whilst using gill netting and electro-fishing. Gill nets trap or wedge fish within the mesh, often causing physical injury to their gills and integument, a method not commonly used if researchers wish to keep fish alive. Spinal injuries caused by muscle contractions can occur whilst electro-fishing which can often lead to fish mortality, though injuries are more common amongst larger fish (Jellyman and Graynoth, 2005). In New Zealand, electro-fishing can be used in freshwaters with depths below 0.75m, though depths greater than this or lowland waterways with saline conditions usually require the use of fyke netting (Jellyman and Graynoth, 2005).

Portt *et al.* (2006) considered that fyke netting was deemed to be a highly effective fish capture technique due to its ability to be set amongst a variety of complex habitat types, including dense macrophytes and coarse woody debris. These nets, which have very little environmental impact, can be easily set in shallow streams, as long as the tunnels and fish holds are fully submerged at all times. The interaction between the stakes and the seabed can cause some disturbance to the habitat, though the impact quickly reduces in tidal areas with a faster flow (Portt *et al.,* 2006).

Due to its light weight and ability to collapse into flat, compact sizes, fyke netting is a convenient and time-efficient method for measuring fish abundance. The equipment is easily hauled by hand and often only requires the use of one researcher to set and collect the nets. The gear has been well adapted to intercept large quantities of small-bodied migrating fish because an attached leader helps guide fish directly into the netting tunnels (Portt *et al.,* 2006). Due to its convenience of use, as well as its minimal effect on fish mortality and habitat destruction, fyke netting is the fish capture method with the fewest trade-offs and is therefore the technique used in this study.

3.2. Sampling

3.2.1. Fish Capture
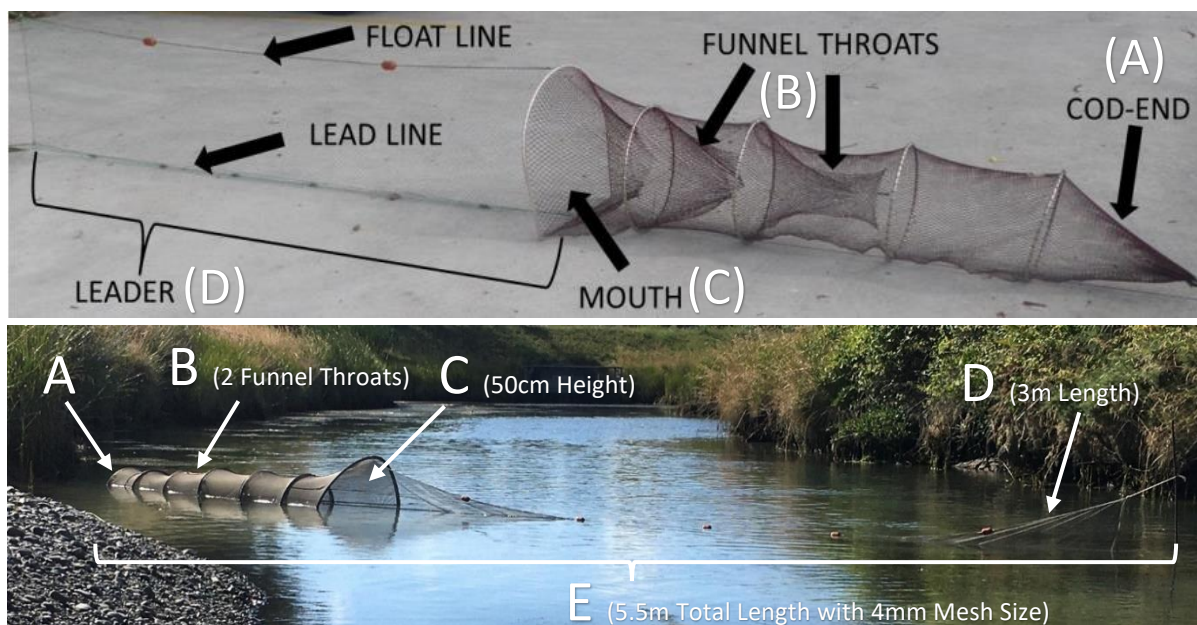
Figure 3.1. Above: Standard fyke net design with labelled elements (Lake, 2013). Below: Fully deployed, 4mm mesh fyke net as used in this study to analyse the abundance and diversity of species along Hawke's Bay's lowland waterways. **A:** The cod-end, or the lower compartment where most caught fish are held, secured by a steel stake. The netting opens at this end to allow efficient removal of caught species. **B:** An attached zip allows access to the net's two funnel-shaped throats. **C:** The D-ring mouth, or the net opening, leading to the inner funnel throats. **D:** Leader with a float line at the top and a lead line at the bottom, used to guide fish into the net opening and also secured by a steel stake. **E:** Total length of the fyke net and leader.

Before entering each waterway, an observational risk assessment of the site was conducted to ensure that any potential dangers were noted and methods were altered, if necessary, with particular attention given to tidal reaches where the water levels were likely to alter throughout the day. Waders were worn in visibly shallow sites and a wet suit with mud shoes were worn in deeper sites, or in sites with poor water clarity. For health and safety reasons, a life jacket was worn when the depth exceeded 1m or when the water flow was noticeably fast. An assistant researcher was present at all times during the surveying in case any additional support was required.

Six fine-meshed (4mm) fyke nets with two funnel throats, excluding the trap mouth, were set in each of the upstream, midstream and downstream sites within each waterway, spanning a total length of 5.5m including the leader. The netting's cod-end (Fig.3.1A) was securely tied and the zip tightly fastened (Fig.3.1B) to prevent caught fish from escaping once inside the net. A steel stake was thread through a rope attached to the net's cod-end and securely embedded within the stream bed, as close to the bank as possible. Any encroaching bank vegetation was carefully removed using a rake, though this was avoided if an alternative suitable netting site was located. The fyke netting was gently lifted and taken downstream along the bank. Each of the net's compartments were widely spread and carefully placed within the waterway, allowing as much of the fyke net to be submerged within the water as possible. For waterways too shallow to allow full net submersion, it was ensured that the mouth, the two funnel throats and the cod-end were at least partially submerged at all times. The 3m long mesh leader (Fig.3.1D) was then spread across the width of the waterway and embedded into the stream bed using a second steel stake. The leader was used to guide migrating

fish into the net opening with the aim of capturing fish which would otherwise bypass the net and continue upstream.

Although directionally selective, the main aim of this study was to assess the impact of barriers on upstream migrating inanga populations, therefore it was essential to ensure that the 50cm high mouth of the net (Fig.3.1C) faced a downstream direction so that fish would encounter the opening whilst migrating upstream, though fish abundance counts may have also included sub-populations which were simply milling about around the nets rather than actively migrating upstream.

The fyke netting was carefully removed from the waterway by firstly unembedding the steel stake that was holding the mesh leader. The leader was then collected, and any plant matter or built-up debris was wiped away. Each of the net's compartments were gradually lifted out of the water, starting from the mouth, whilst being carefully shaken to allow the contents of the net to pass down through the funnel throats and into its cod-end. The second steel stake was removed, though the cod-end remained submerged within the water.
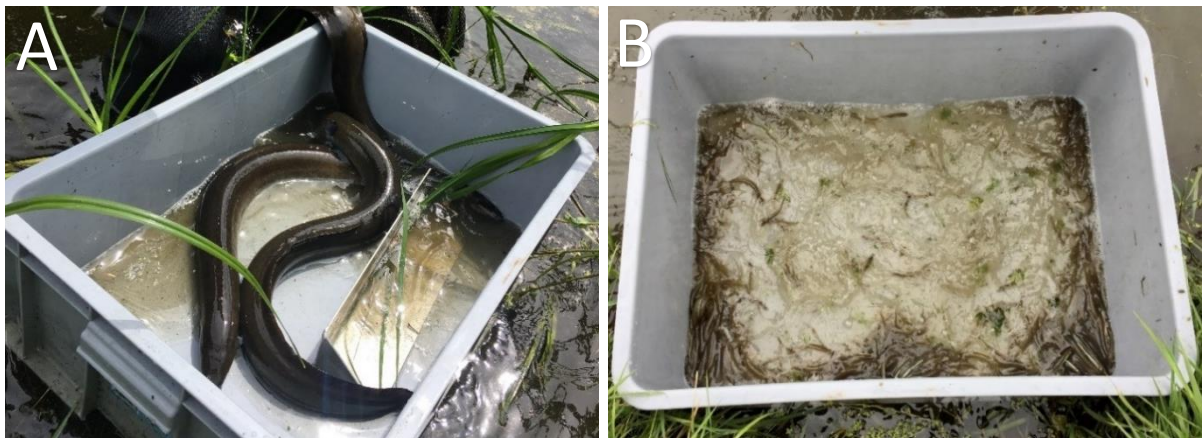
Figure 3.2. A: Caught large-bodied species like the shortfin eel were firstly removed from the netting through the zip compartment and carefully placed into a measuring box. **B:** Inanga and other caught small-bodied species are then placed into the box and their abundance counted.

The central tunnel's zip was unfastened, and the largest sized caught species were gently removed into a plastic measuring box which was holding some stream water. The initial catch was mainly comprised of shortfin eels *(Anguilla australis)* (Fig.3.2A) and New Zealand longfin eels *(Anguilla dieffenbachia)* which were identified and swiftly released into the same region as they were caught. The net's cod-end was lifted from the water, untied and its contents released into the measuring box (Fig.3.2B) which again contained fresh stream water. Fish were identified down to a species level and their abundance was tallied. Once an individual had been counted, it was carefully released back into the waterway close to where the fyke net had been set. After all individuals had been counted, the fyke netting was removed from the waterway and placed onto the bank's edge, though it was ensured that no surrounding habitat was disturbed. After each site, nets were checked for damage, sterilised with bleach and left to dry before being stored, ready for their next use.

Six fine-meshed (4mm) fyke nets were set approximately 15m apart on alternating stream banks, totalling three nets on both of the true left and the true right banks and spanning a ~90m total length across the stream to ensure that as few fish as possible were able to bypass all of the nets. Nets were set at a 45∘ angle to the bank with the cod-end securely anchored as close to the bank as possible, and the mesh leader reached across the stream at an approximate 90∘ angle and was secured into the stream bed. In tidal sites, nets were set along areas within the stream where a low

tide would still at least partially submerge the cod-end and the funnel throat elements, though nets that were set within areas with a considerably strong water flow or those considered too deep for efficient fish capture were disregarded from the survey and subsequent analyses. For sites with upstream and downstream sub-sites, a total of 12 nets were set, and for sites with the additional midstream sub-site, a total of 18 nets were set, unless stated otherwise in the specific site descriptions. Nets were left for four hours to allow sufficient time for migrating fish to encounter and willingly enter the netting.

3.2.2. Sampling Design

The analyses of the abundance of inanga and the total abundance of all species combined has each been assessed using the independent variables 'Barrier Status' and 'Treatment', resulting in four individual datasets titled:

'Inanga Abundance Vs Barrier Status'
'Inanga Abundance Vs Treatment'
'All Species Total Abundance Vs Barrier Status'
'All Species Total Abundance Vs Treatment'

The 'Barrier Status' variable has three levels, each with either 2 or 3 of the 'Upstream (US)', 'Midstream (MS)' and 'Downstream (DS)' 'positions':

'Control' – Upstream (US), Midstream (MS), Downstream (DS)
'Barrier' – Upstream (US), Downstream (DS)
'Non-Barrier' – Upstream (US), Midstream (MS), Downstream (DS)

Waterways categorised as a 'control' have no drainage structure present. There is a drainage structure present along waterways categorised under the 'barrier' level and these structures do act as a complete, permanent barrier to upstream migrating fish. There is also a drainage structure present along waterways categorised as a 'non-barrier', though these structures do not act as a full barrier to upstream migrating fish and allow at least partial upstream passage.

The 'Treatment' variable has six levels, each also with either 2 or 3 of the 'Upstream (US)', 'Midstream (MS)' and 'Downstream (DS)' 'positions':

'Control' – Upstream (US), Midstream (MS), Downstream (DS)
'Bridge Apron' – Upstream (US), Downstream (DS)
'Culvert' – Upstream (US), Downstream (DS)
'Flood Gate' – Upstream (US), Downstream (DS)
'Pumping Station' – Upstream (US), Downstream (DS)
'Pumping Station / Fish-Friendly Flood Gate (PS/FFFG)' – Upstream (US), Midstream (MS), Downstream (DS)

For waterways with a 'Barrier Status' or a 'Treatment' level *not* categorised as a 'control', the upstream position is defined as a sub-site above a drainage structure and the downstream position is defined as a sub-site below a drainage structure. The midstream position occurs when two drainage structures are present along a single waterway and is defined in this study as a sub-site which is below the most upstream structure and is above the most downstream structure. For 'control' waterways where no drainage structure exists, overhead bridges which do not act as migration barriers were used to divide upstream, midstream and downstream positions along a

waterway. The assumptions or local knowledge that inanga inhabit these particular areas and the ease of access to each of the sub-sites also influenced the location of each of the three positions along 'control' waterways.

3.2.3. Habitat and Water Chemistry Sampling Methods

To examine the habitat use and the effect of the environment's temperature, dissolved oxygen and specific conductivity measures on the inanga abundance and on the total abundance of all species combined, assessments and surveys were conducted in the upstream, the midstream and the downstream sub-sites along each of this study's 16 waterways.

The depth of each sub-site was also measured by dividing the waterway into three transects which encompassed all six of the set fyke net locations. The first transect started at the point at which the most upstream fyke net was set, the second was located between fyke nets three and four and the third transect lay where the sixth, or the most downstream fyke net was set. The second transect location was altered accordingly if fewer than six nets were set. Five evenly spaced depth measures were taken across the width of the channel along each transect , perpendicular to the stream bank using a measuring pole. The mean depth of each transect was calculated in centimetres and was used to calculate a mean total depth for the entire sub-site.

To ensure consistency within the results, no in situ measurements of the channel widths were taken as a number of waterways were considered too wide to obtain reliable measures. The mean width of each sub-site was therefore calculated in metres using the online Google Earth measuring tool (Google Earth, 2021). Photographs of each sub-site's habitat and drainage structure, if present, were taken for future observational comparisons.

After calibrating a YSI metre with clean sterile water, a water temperature (°C), dissolved oxygen (%) and specific conductivity (SPC) (μS/cm) reading was collected at 9am each morning and again at 1pm each afternoon in the same location along each sub-site to account for varying climatic conditions throughout the day. The YSI metre's sensor probe was fully submerged into the waterway around the margins of each water channel, away from any macrophytes or debris. Readings were taken at the banks of the second transects, or around the most midstream section of each sub-site, in order to yield the most representative results of this study's fish sampling zone. Probes were left submerged for five minutes at a depth of 20cm from the water's surface along all sub-sites to ensure consistency across the readings and were submerged away from each fyke net's mouth element to minimise any disturbance caused to nearby fish. The morning and the afternoon water quality readings were then used to calculate a mean daily temperature, dissolved oxygen and SPC measure for each waterway's sub-sites to be used in this study's analyses.

3.3. Ethical Considerations

Although the data used in this Masters of Research study was collected as part of a separate piece of work and therefore no ethical rules or regulations of Bangor University could be followed, a number of ethical issues were still considered before and during the data collection process.

Fyke netting is considered a passive and non-destructive fish capture method, though it remained important to consider where the nets were set to ensure the survival of the caught fish. It was essential that the net's tunnel floors, and cod-end remained submerged within the water during

both high and low tide times along tidal reaches to prevent fish suffocation. Whilst counting the abundance of caught fish, larger bodied individuals were tallied and released first to reduce the risk of suffocation, as well as reducing stress levels for smaller bodied species. A sufficient volume of water was used within the measuring box at all times to ensure that all individuals had space to freely swim within the box. Once tallied, each fish was gently released into the same region as they were initially caught with the aim of reducing the impact on their upstream migrations.

3.4. Potential Obstacles
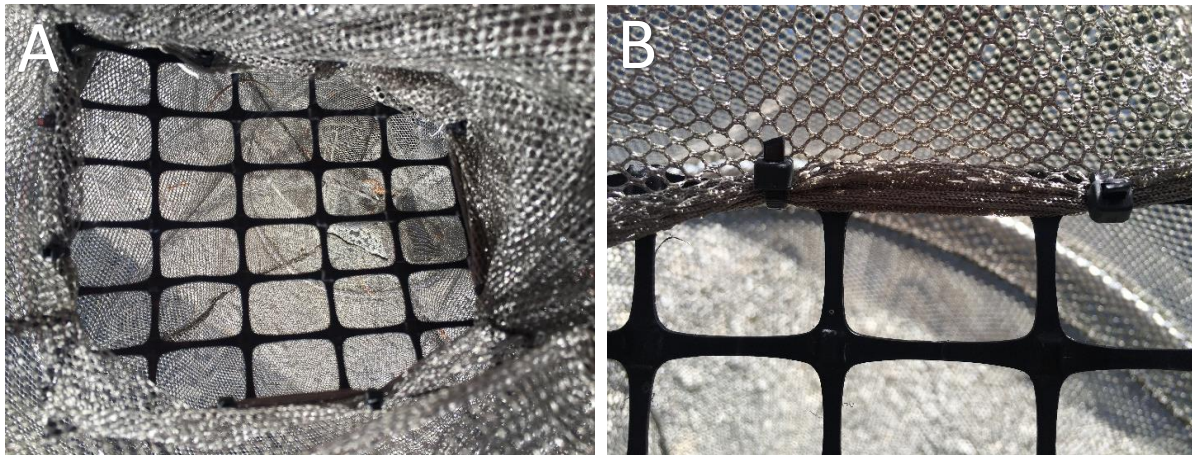
Figure 3.3. **A:** Plastic exclusion grills with 4cm wide gaps were attached onto each of the fyke netting's two funnel throats. **B:** The grills were tightly secured using cable ties which were then cut down using pliers.

Although the benefits of fyke netting as a fish capture method are considerable, a number of potential obstacles may arise which will need remediating. An unnatural accumulation of small-bodied prey species in the cod-end of the net may attract predators like eels or larger bodied fish causing predation and high fish mortality. Not all individuals will then be included in the abundance count and the data will be unreliable. To avoid this potential obstacle, plastic exclusion grills with 4cm x 3cm sized gaps were cut to shape using pliers and placed around each of the two funnel throats within the fyke net (Fig.3.3A) having been secured into place using cable ties (Fig.3.3B). The 4cm x 3cm sized gaps in the grills were large enough in size to allow full passage of small-bodied species into the cod-end of the net but were too small to allow passage of large predators, like eels or adult grey mullet *(Mugil cephalus).*

Fyke netting can be used in a variety of habitat types, though setting the nets may prove difficult within the deepest waterways or along sites with excessive macrophyte density. Extensive researching of the site and careful planning of where to set nets were required pre-data collection, to avoid areas with the most unsuitable environmental conditions. However, this was not always sufficient, and on some occasions, true conditions were observed on the day of data collection only. Excessive submerged macrophytes were removed using a rake to provide enough space for nets to be set. If a region of a sub-site was deemed to be too deep to set fyke netting, all surrounding areas were thoroughly examined to determine whether they could be used at an alternative location. If a substitute sub-site was not found, as many nets were set in the original sub-site as possible.

3.5. Data Processing

Prior to statistical analyses and model justification, all datasets were inspected to ensure that no values were missing and that all species counts where no individuals were caught were filled with a zero. This study's quantitative count data varies by large margins; zeros account for ~80% of the data whilst a few considerably larger abundance results warped much of the remaining data. Therefore, it was decided that the data would be transformed based on log transformations [ln(N+1)] for graphical presentations only, whilst specialised statistical models were used to account for these issues in the analyses discussed below.

Before analysing quantitative and discrete data, a correct statistical model must be chosen which accounts for the high levels of overdispersion and excess zeros which are commonly associated with count data. Using RStudio version 3.5.1, the statistical distributions:

Poisson

with the formula:

$$P(X) = (e^{-\lambda}) \, (\lambda^x) \, / x!$$

Where:
λ = The mean number of successes that occur in an interval,
x = The actual number of successes that occur in an interval

negative binomial (Neg Bin)

with the formula:

$$f(x) = [\, (s+x-1) \, / x \,] \, p^s \, (1-p)^x$$

Where:
x = The total number of trials,
s = The number of occurrences of success,
p = The probability of success on each occurrence,
1-p = The probability of failure on each occurrence

zero-inflated Poisson (ZIP)

with the formula:

$$f_{ZI}(0) = (1-\pi) + \pi f(0)$$
$$f_{ZI}(x) = \pi f(x) \qquad\qquad x = 1, 2, 3,\ldots$$

Where:
$f(x)$ = The above Poisson formula

zero-inflated negative binomial (ZINB)

with the formula:

$$f_{ZI}(0) = (1-\pi) + \pi f(0)$$
$$f_{ZI}(x) = \pi f(x) \qquad x = 1, 2, 3,...$$

Where:
$f(x)$ = The above negative binomial formula

...were assessed and compared to test which was the most appropriate model to use to analyse the significance between the dependent variables: 'Inanga Abundance' and 'All Species Total Abundance' in this study's abundance and environmental analyses. These comparisons were conducted for each of the following four datasets:

'Inanga Abundance Vs Barrier Status'
'Inanga Abundance Vs Treatment'
'All Species Total Abundance Vs Barrier Status'
'All Species Total Abundance Vs Treatment'

Initial statistical model comparisons concentrated on the use of the Poisson generalised linear model (GLM) for each dataset in the abundance analyses and a Poisson generalised linear mixed model (GLMM) in the environmental analyses. Quantile residual QQ plots were generated with one-sample Kolmogorov-Smirnov (KS), outlier and dispersion tests, where significant results indicate that the Poisson model should be rejected in favour of an alternative, better fitting model. Visual assessment of the distribution of observed residuals across the QQ plot indicate whether the Poisson is a significantly appropriate model to use for that particular dataset, with an accumulation of observed residuals around the tail ends of the plot indicating that a high level of overdispersion exists.

Additional Pearson-chisq parametric dispersion tests confirm whether overdispersion occurs under the model as values greater than 1 indicate that true overdispersion exists, whilst zero-inflation tests yield ratios of observed to predicted zeros to assess whether the model is underfitting the number of zeros in the dataset, where ratios greater than 1 show that zero-inflation exists. If so, negative binomial and zero-inflated distributions were analysed with the same tests to assess whether each model rectified these issues and whether they were a significantly better fit to use for each dataset.

Further model comparisons via Akaike Information Criterion (AIC) were conducted for each dataset across the abundance and the environmental analyses, where the lowest AIC value indicates a more parsimonious and better fitting model.

Additional likelihood-ratio tests (LRTs) and Vuong tests were conducted and analysed (Table 3.1; Table 3.2) for the abundance analyses only. LRTs are used to compare nested models, like the Poisson Vs the negative binomial, or the zero-inflated Poisson Vs the zero-inflated negative binomial. A log-likelihood value is generated which summarises the fit of the observed to the expected values for each model. The difference is these values is assessed against the chi-squared distribution, which if significant ($p < 0.05$), indicates which model is the most suitable to use (Table 3.1). For non-nested model comparisons, like the negative binomial Vs the zero-inflated negative binomial, Vuong tests were used which generates and compares Vuong AIC test statistics for each dataset, indicating which of the two models is the most significantly appropriate to use (Table 3.2).

3.5.1. Justification of Models Used in the Abundance Analyses

3.5.1.1. Inanga Abundance Vs Barrier Status

A poorly fitting Poisson GLM:

Significant (p = 0) KS, outlier and dispersion tests = the Poisson model should be rejected.
A large number of observed residuals accumulated around the tail ends of the distribution on a QQ plot than would be expected under a fitted model.
Pearson-chisq dispersion value of 44.45 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 2.11 (p < 2.2E-16) = zero-inflation.

Significant overdispersion and zero-inflation exists under a Poisson distribution, warranting the assessment of a negative binomial model, which unlike the Poisson, does not assume that the mean is equal to the variance and has an additional dispersion parameter, theta (θ).

A more suitable, better fitting negative binomial GLM:

Non-significant KS (p = 0.90), outlier (p = 1) and dispersion tests (p = 0.99) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq dispersion value of 0.72 (p = 1) = overdispersion does not exist. Test values of < 1 indicate that the data is underdispersed, though the good fit of the model on the residual QQ plot suggests that underdispersion of this dataset is not as issue under the negative binomial distribution.
Zero-inflation ratio of 1.00 (p = 1) = zero-inflation does not exist.

Further AIC, LRT (Table 3.1) and Vuong tests (Table 3.2) confirm that the **negative binomial GLM** is the most appropriate model to use in the abundance analysis of the 'Inanga Abundance Vs Barrier Status' dataset.

3.5.1.2. Inanga Abundance Vs Treatment

A poorly fitting Poisson GLM:

Significant (p = 0) KS, outlier and dispersion tests = the Poisson model should be rejected.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 43.94 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 1.67 (p < 2.2E-16) = zero-inflation.

A more suitable, better fitting negative binomial GLM:

Non-significant KS (p = 0.87), outlier (p = 1.00) and dispersion tests (p = 0.84) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq dispersion value of 0.83 (p = 0.95) = overdispersion does not exist.
Zero-inflation ratio of 1.00 (p = 1) = zero-inflation does not exist.

Further AIC, LRT (Table 3.1) and Vuong tests (Table 3.2) confirm that the **negative binomial GLM** is the most appropriate model to use in the abundance analysis of the 'Inanga Abundance Vs Treatment' dataset.

3.5.1.3. All Species Total Abundance Vs Barrier Status

A poorly fitting Poisson GLM:

Significant (p = 0) KS, outlier and dispersion tests = the Poisson model should be rejected.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 71.36 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 7.68 (p < 2.2E-16) = zero-inflation.

A more suitable, better fitting negative binomial GLM:

Non-significant KS (p = 0.14), outlier (p = 0.19) and dispersion tests (p = 0.37) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq statistic of 1.23 (p = 3.17E-11) = overdispersion does not exist.

Further AIC, LRT (Table 3.1) and Vuong tests (Table 3.2) confirm that zero-inflation remains under a negative binomial distribution and that the **zero-inflated negative binomial GLM** is the most appropriate model to use in the abundance analysis of the 'All Species Total Abundance Vs Barrier Status' dataset.

3.5.1.4. All Species Total Abundance Vs Treatment

A poorly fitting Poisson GLM:

Significant (p = 0) KS, outlier and dispersion tests = the Poisson model should be rejected.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 70.30 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 5.65 (p < 2.2E-16) = zero-inflation.

A more suitable, better fitting negative binomial GLM:

Non-significant KS (p = 0.81), outlier (p = 0.11) and dispersion tests (p = 0.39) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq dispersion value of 1.29 (p = 4.6E-16) = a small amount of overdispersion exists, though considerably less than under a Poisson distribution.

Further AIC, LRT (Table 3.1) and Vuong tests (Table 3.2) confirm that zero-inflation remains under a negative binomial distribution and that the **zero-inflated negative binomial GLM** is the most appropriate model to use in the abundance analyses of the 'All Species Total Abundance Vs Treatment' dataset.

Poisson and negative binomial GLM QQ plots for each of the four datasets, along with zero-inflation charts of the frequency of simulated zeros and rootograms are available in Appendix A.

3.5.1.5. AIC, LRT and Vuong Comparisons

Table 3.1. AIC comparisons and Likelihood-Ratio Tests (LRTs) between the nested Poisson and the negative binomial (Neg Bin), and the zero-inflated Poisson (ZIP) and the zero-inflated negative binomial (ZINB) GLMs for the 'Inanga Abundance Vs Barrier Status', 'Inanga Abundance Vs Treatment', 'All Species Total Abundance Vs Barrier Status' and 'All Species Total Abundance Vs Treatment' datasets. AIC = Akaike Information Criterion, Chisq = Pearson's Chi-squared statistic, df = degrees of freedom. Values in **bold** represent those that have a significant p-value of <0.05.

Dataset	Statistical Model Comparison					
	Poisson	Vs	Neg Bin	ZIP	Vs	ZINB
Inanga Abundance Vs Barrier Status						
AIC	5279.61		864.81	3732.05		873.39
-2log-likelihood	5263.6		846.82	3700.04		839.38
Chisq			**4416.8**			**2860.7**
df	8		9	16		17
Inanga Abundance Vs Treatment						
AIC	5018.86		853.20	3696.52		869.81
-2log-likelihood	4990.8		823.2	3640.6		811.8
Chisq			**4167.7**			**2828.7**
df	14		15	28		29
All Species Total Abundance Vs Barrier Status						
AIC	34726.53		4795.67	16829.36		4757.52
-2log-likelihood	34710.6		4777.6	16797.4		4723.6
Chisq			**29933**			**12074**
df	8		9	16		17
All Species Total Abundance Vs Treatment						
AIC	34221.65		4784.17	16280.91		4726.38
-2log-likelihood	34193.6		4754.2	16225		4668.4
Chisq			**29439**			**11557**
df	14		15	28		29

Table 3.2. Vuong tests between the non-nested negative binomial (Neg Bin) and the zero-inflated negative binomial (ZINB) models for the 'Inanga Abundance Vs Barrier Status', 'Inanga Abundance Vs Treatment', 'All Species Total Abundance Vs Barrier Status' and 'All Species Total Abundance Vs Treatment' datasets. AIC = Akaike Information Criterion. Significantly better fitting models, where p < 0.05, are in **bold.**

Dataset	Neg Bin Vs ZINB Comparison	
	Vuong AIC Test Statistic	Preferable Model
Inanga Abundance Vs Barrier Status	1.72	**Neg Bin**
Inanga Abundance Vs Treatment	2.64	**Neg Bin**

All Species Total Abundance Vs Barrier Status	-2.59	**ZINB**
All Species Total Abundance Vs Treatment	-3.29	**ZINB**

3.5.1.6. Visual Representations

Box and whisker plots were produced, showing the transformed inanga and all species abundance data against position along waterways categorised by barrier status and treatment. A box and whisker plot shows the descriptive statistics for each covariate. The median value of the data is represented by a thick black line running through the box, whilst the box itself represents the interquartile range. The whiskers, or the black lines stretching from the top and the bottom ends of the box represent the minimum and the maximum values. Additional jitter plots, represented by scattered circular points on the graphs were included over each box and whisker bar, illustrating the distribution of the data and showing overlapping data points more clearly than a standard strip plot.

For illustrative purposes only, raw data was log transformed using the ln(N+c) function, where N = abundance and c, a constant, was equal to 1 for all datasets in this study. Transformations were plotted against position, barrier status and treatment on box and whisker plots and were used only to improve the interpretability of the data, preventing skewness towards values which were larger than much of the remaining data points.

However, raw abundance data that had not been transformed was used for the statistical analyses of all datasets. A large amount of literature, including a paper by O'Hara and Kotze (2010) showed that transformed variables often perform poorly, produce a large amount of bias and can lead to impossible predictions, like negative abundances, in comparison to GLMs which take into account overdispersion and excess zero issues. They further recommend that count data analyses should exclude any transformations and that GLMs should be used instead.

The position, barrier status and treatment independent variables and their interaction terms predicted the negative binomial count model for the two inanga datasets and the count and zero-inflated logistic components of a zero-inflated negative binomial model for the two all species datasets. Regression coefficients, p-values and standard errors were estimated for each dataset, with abundances in the downstream and the control groups being used as the reference groups and noted as the '(Intercept)' covariate. A significant p-value level of < 0.05 was used across this study.

(Zero-inflated) negative binomial models yield natural log units, which can be difficult to interpret in a real-world setting. Therefore, 'Incidence Rate Ratios' (IRR) and 95% confidence intervals (CI) based on maximum likelihood estimates (MLE) were calculated by raising the base constant e to the power of the count model coefficient, or in other words, the regression coefficients were exponentiated. The IRR shows the increase in the number of predicted counts at upstream and midstream positions against the downstream intercept along waterways categorised by barrier status and treatment, given all other factors were held constant. 'Odds Ratios' (OR) were calculated by exponentiating the coefficients of the logistic component of zero-inflated models and were used to estimate the odds of there being an excess zero within the dataset. IRR and OR values greater than 1 suggest a higher probability of the event occurring than the reference group; values less than 1 suggest a lower probability.

3.5.2. Description of Models Used in the Biodiversity Analyses

To quantify and compare the diversity and the evenness of species between upstream (US), midstream (MS) and downstream (DS) sub-sites along waterways with varying barrier status or treatment, the Shannon Weiner diversity index (H') and Pielou's evenness index (J') were calculated. The number of individuals caught per species (N_i) and the grand total number of individuals caught of all species (T_i) were pooled across all set fyke nets within a given sub-site to yield total abundances and species richness measures. These were then compared between the 16 sampled waterways, the three barrier status levels and the six treatment levels as described in Table 2.1 and were used in the following biodiversity, evenness and similarity analyses.

The Shannon Weiner index measures the number of species within a sub-site and their relative abundance, and is calculated using the following formula:

$$H' = -\sum [\, (p_i) \times \ln(p_i) \,]$$

Where:

$p_i = N_i / T_i$,
N_i = Abundance of a single species in the sub-site,
T_i = Total abundance of all species in the sub-site.

The 'Effective Number of Species' (ENS) was calculated as an additional statistical measure, showing the true diversity of a sub-site associated with the Shannon Weiner index, if all species are present in equal abundances:

$$ENS = EXP(H')$$

Pielou's evenness index (J') measures how similar the abundances of different species are between sub-sites with values ranging between 0 and 1, where 0 values represent no evenness between sub-sites and 1 represents complete evenness between sub-sites. J' is calculated using the formula:

$$J' = H' / H'_{max}$$

Where:

$$H'_{max} = \ln(S)$$

S = Species richness of a sample, which for this study ranges between 0 species and a maximum of 10 species.

The Sørensen's similarity index (ß) was also used to compare the similarity of species between sub-sites and was calculated using the following formula:

$$ß = 2C / (S1 + S2)$$

Where:

C = The number of shared species between S1 and S2,
S1 = Species richness in sub-site 1,
S2 = Species richness in sub-site 2.

A ß value of 0 means represents two sub-sites which do not share any species, whereas a ß value of 1 shows that the exact same species are present within both sub-sites. ß values can then be multiplied by 100 to yield percentage similarity scores.

Further information regarding the common and Latin names, short codes, life histories and the New Zealand status of the 10 species present across the sampled sites of this study is available in Appendix A: Species Diversity and Similarity (7.1.3; Table 7.1).

To further compare the biodiversity within two sub-sites, a Hutcheson t-test *(t)* was conducted which uses the variance of the Shannon Weiner diversity index to obtain a statistical significance of the difference between the two samples. P-values were calculated and used to either reject or fail to reject the null hypothesis that biodiversity will not differ between sub-sites if fish passage is unrestricted. Unlike other hypotheses tests, the Hutcheson t-test does not require replicated data for statistical analysis and is not dependent upon the sample size and is therefore the preferred method for comparing H' values. The Hutcheson t-test compared H' values between the US and the MS, or the US and the DS sub-sites for waterways with a control, barrier or a non-barrier status and for waterways with a control, bridge apron, culvert, flood gate, pumping station or a PS/ FFFG treatment, rather than comparing H' values for individual status or treatments against the H' values of the control sites. The Hutcheson t-test was calculated using the following formula:

$$t = H'1 - H'2 / \sqrt{(S^2_{H'1} + S^2_{H'2})}$$

Where:

H'1 = The Shannon Weiner diversity index of sub-site 1,
H'2 = The Shannon Weiner diversity index of sub-site 2,
$S^2_{H'1}$ = The variance of H'1,
$S^2_{H'2}$ = The variance of H'2.

A significant p-value level of < 0.05 was used across this study.

3.5.3. Justification of Models Used in the Environmental Analyses

Unlike GLMs which incorporate fixed effects only, GLMMs use both fixed and random effects. In the following analyses, the environmental variables: temperature, dissolved oxygen and specific conductivity acted as the fixed effects, or factors which were of particular interest. In order to avoid any convergence or singular model warnings, the three environmental variables were scaled and centred using scale functions in R. Scaling, or standardisation of the data, is useful whilst comparing independent variables with scales of different magnitudes like those in this study.

However, due to the high overparameterization which occurs whilst running a GLMM with data with small sample sizes, statistical analyses of the effect of the environmental variables on the inanga and the all species total abundances along individual, site-specific waterways could not be reliably calculated. Therefore, as in the abundance and the biodiversity analyses, sites were categorised by barrier status and treatment (Table 2.1).

To account for the non-independence of the environmental variables across the data, barrier status, treatment and position were included as random effects. The position variable was nested within the barrier status and the treatment variables, i.e., each barrier status and treatment level included US, (MS) and DS positions. An intercept for the random effects was included to allow for multiple, differing responses based specifically on each position, barrier status and treatment level.

Poisson and negative binomial GLMMs with scaled fixed effects and nested random effects were run and compared to assess which was the better fitting and the most appropriate model to use in the analysis of each of the four datasets.

3.5.3.1. Inanga Abundance Vs Barrier Status

A poorly fitting Poisson GLMM (AIC = 4265.13):

Significant KS test (p = 0) = the Poisson model should be rejected, although outlier (p = 1) and dispersion (p = 0.66) tests are non-significant.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 35.94 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 1.63 (p = 0.18) = zero-inflation.

A more suitable, better fitting negative binomial GLMM (AIC = 876.49):

Non-significant KS (p = 0.61), outlier (p = 1.00) and dispersion tests (p = 0.44) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq dispersion value of 0.89 (p = 0.85) = overdispersion does not exist.
Zero-inflation ratio of 1.02 (p = 0.92) = zero-inflation does not exist.

As overdispersion and zero-inflation do not exist, no further statistical model comparisons are required. With the lowest AIC value, the **negative binomial GLMM** is the most appropriate model to use in the environmental analysis of the 'Inanga Abundance Vs Barrier Status' dataset.

3.5.3.2. Inanga Abundance Vs Treatment

A poorly fitting Poisson GLMM (AIC = 4142.91):

Significant KS test (p = 0.046) = the Poisson model should be rejected, although outlier (p = 1) and dispersion (p = 0.66) tests are non-significant.
A slight accumulation of observed residuals around the tail ends on a QQ plot.
Pearson-chisq dispersion value of 32.97 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 1.30 (p = 0.35) = zero-inflation.

A more suitable, better fitting negative binomial GLMM (AIC = 877.99):

Non-significant KS (p = 0.09), outlier (p = 1.00) and dispersion tests (p = 0.30) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq dispersion value of 0.74 (p = 1.00) = overdispersion does not exist.
Zero-inflation ratio of 0.93 (p = 0.72) = zero-inflation does not exist.

As overdispersion and zero-inflation do not exist, no further statistical model comparisons are required. With the lowest AIC value, the **negative binomial GLMM** is the most appropriate model to use in the environmental analysis of the 'Inanga Abundance Vs Treatment' dataset.

3.5.3.3. All Species Total Abundance Vs Barrier Status

A poorly fitting Poisson GLMM (AIC = 32252.75):

Significant KS (p = 0), outlier (p = 0) and dispersion (p = 0.03) tests = the Poisson model should be rejected.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 53.15 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 3.20 (p < 2.2E-16) = zero-inflation.

A more suitable, better fitting negative binomial GLMM (AIC = 4770.66):

Non-significant KS (p = 0.30), outlier (p = 1.00) and dispersion tests (p = 0.86) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq statistic of 1.20 (p = 8.57 E-08) = overdispersion does not exist.
Zero-inflation ratio of 0.99 (p = 0.79) = zero-inflation does not exist.

As overdispersion and zero-inflation do not exist, no further statistical model comparisons are required. With the lowest AIC value, the **negative binomial GLMM** is the most appropriate model to use in the environmental analysis of the 'All Species Total Abundance Vs Barrier Status' dataset.

3.5.3.4. All Species Total Abundance Vs Treatment

A poorly fitting Poisson GLMM (AIC = 32220.80):

Significant (p = 0) KS, outlier and dispersion tests = the Poisson model should be rejected.
An accumulation of observed residuals around the tail ends of the distribution on a QQ plot.
Pearson-chisq dispersion value of 53.00 (p < 2.2E-16) = overdispersion.
Zero-inflation ratio of 2.95 (p < 2.2E-16) = zero-inflation.

A more suitable, better fitting negative binomial GLMM (AIC = 4776.12):

Non-significant KS (p = 0.14), outlier (p = 1.00) and dispersion tests (p = 0.85) = fail to reject the negative binomial model.
No significant differences between the observed and expected residuals on a QQ plot.
Pearson-chisq statistic of 1.20 (p = 4.38 E-09) = overdispersion does not exist.
Zero-inflation ratio of 0.98 (p = 0.30) = zero-inflation does not exist.
As overdispersion and zero-inflation do not exist, no further statistical model comparisons are required. With the lowest AIC value, the **negative binomial GLMM** is the most appropriate model to use in the environmental analysis of the 'All Species Total Abundance Vs Barrier Status' dataset.

Poisson and negative binomial GLMM QQ plots for each of the four datasets, along with zero-inflation charts of the frequency of simulated zeros are available in Appendix A.

3.5.3.5. Visual Representations

Although categorising individual sites by barrier status and treatment rectified the overparameterization issues in the statistical analyses as described above, issues remained whilst attempting to visualise the results. As only one waterway was sampled for each of the bridge apron, the culvert and the PS/ FFFG treatments, only one mean temperature, one mean dissolved oxygen saturation and one mean specific conductivity measure was available for each sub-site position across these treatments. Therefore, reliable comparisons using the raw inanga and all species total abundance data could not be made. In order to overcome these issues, predicted abundance counts were calculated and used to reliably compare the estimated inanga and the all species total abundances for each barrier status and treatment level.

Estimated Marginal Means (EMMs) were calculated by running negative binomial GLMMs within the ggpredict() command in R. This yielded predicted inanga abundances and predicted all species total abundances for each of the upstream, the midstream and the downstream positions along waterways categorised by the random effects: barrier status and treatment. Predicted abundances were plotted against the fixed and interacting environmental effects: mean temperature (°C), mean dissolved oxygen saturation (%) and mean specific conductivity (μS/cm).

Upper and lower 95% intervals were not included on the EMM plots to allow clearer visualisations and more precise comparisons of each data level.

Random Effects plots of the Incidence Rate Ratios (IRRs) and 95% confidence intervals of the nested position, barrier status and treatment random effects are available in Appendix A.

Chapter 4

Results

4.1. Data Description for Abundance and Biodiversity Analyses

The abundance data used in this study refers to the total number of individuals (N) per species caught within each of the upstream, the midstream and the downstream sub-sites along 16 of Hawke's Bay's lowland waterways. The number of caught individuals (N) per species and the total number of caught individuals of all species were pooled across all set fyke nets within a given sub-site to give the total number of caught individuals per species and the grand total number of individuals caught of all species as reported in the tables below. The inanga abundance analyses compares the abundance of inanga only (noted as the "inga" short code in Tables 4.1, 4.2 and 4.3) between sub-sites across waterways varying by site (Table 4.1), barrier status (Table 4.2) and treatment (Table 4.3). The all species total abundance analyses compares the grand total number of caught individuals of all species, including inanga, between sub-sites across waterways varying by site (Table 4.1), barrier status (Table 4.2) and treatment (Table 4.3).

As the all species total abundance includes species with differing ecologies and swimming capabilities, additional biodiversity and species richness statistical analyses were conducted to assess whether the species composition of each sub-site differs according to site, barrier status or treatment. The number of caught individuals (N) per species, the grand total number of individuals caught, and the species richness of each site (Table 4.1), barrier status (Table 4.2) or treatment (Table 4.3) were used during the biodiversity, species evenness and species similarity analyses in 'Results section 4.3.'.

Each of the 16 waterways sampled in this study have been categorised by barrier status and treatment (Table 2.1), the two main independent variables used in the analyses in this study.

The differences between inanga abundances, the all species total abundances and the biodiversity and species richness between sub-sites have also been statistically analysed for each of the 16 sampled waterways to enable site-specific management solutions to any potential migratory barriers. These analyses can be found in detail in Appendix B.

Table 4.1. The total number of caught individuals (N) per species, including the focal inanga, the grand total number of individuals caught and the species richness for the upstream (US), midstream (MS) and downstream (DS) sub-sites for each of the 16 sampled lowland waterways in this study.

Site Short Code and Position	Total Number of Caught Individuals (N) per Species (Short Code)										Grand Total	Species Richness
	inga	cmbu	crab	gamb	gldf	grml	leel	seel	srmp	ylml		
taup												
US	124	1	0	10	0	0	0	0	0	0	135	3
DS	657	0	0	14	3	0	0	4	0	0	678	4

talm												
US	28	0	1	91	7	0	0	21	0	0	148	5
DS	71	0	2	30	0	6	0	10	61	21	201	7
grcr												
US	462	1	5	0	0	0	0	0	0	0	468	3
DS	28	2	8	5	0	2	0	0	204	0	249	6
tnst												
US	6	108	3	4	0	0	0	0	0	0	121	4
DS	0	0	0	7	0	0	1	1	30	0	39	4
past												
US	41	1	0	0	0	0	0	1	0	0	43	3
MS	9	0	1	0	0	0	0	0	0	0	10	2
DS	149	64	12	2	0	0	1	0	127	2	357	7
mgst												
US	1	13	0	0	0	0	1	0	0	0	15	3
DS	5	18	0	0	0	0	0	2	0	0	25	3
ngbw												
US	109	3	2	20	0	0	0	0	70	25	229	6
DS	9	24	11	7	0	0	0	0	16	27	94	6
twst*												
US	-	-	-	-	-	-	-	-	-	-	-	-
DS	-	-	-	-	-	-	-	-	-	-	-	-
paps												
US	0	0	0	21	0	0	1	1	0	0	23	3
DS	0	17	0	2	0	0	0	0	0	27	46	2
mpdr												
US	0	0	0	1	0	0	0	0	824	0	825	2
DS	28	2	8	5	0	2	0	0	204	0	249	6
mucr												
US	0	0	0	0	28	0	0	0	0	0	28	1
MS	286	15	0	383	6	0	0	4	0	2	696	6
DS	13	0	15	5	0	5	0	0	111	2	151	6
codr												
US	0	0	0	28	0	0	1	35	0	0	64	3
DS	102	6	0	7	0	0	0	0	0	8	123	4
otri												
US	0	0	0	104	0	0	0	6	0	0	110	2
DS	0	0	0	31	0	0	2	4	0	0	37	3
pust												
US	0	0	0	215	38	0	0	2	0	0	255	3
DS	71	0	2	30	0	6	0	10	61	21	201	7
asdr												
US	0	21	0	816	0	0	0	2	42	0	881	4
DS	71	0	2	30	0	6	0	10	61	21	201	7
pldr												
US	0	0	0	20	0	0	0	18	0	0	38	2
DS	3	1	0	8	0	0	1	2	31	13	59	7

Table 4.2. The total number of caught individuals (N) per species, including the focal inanga, the grand total number of individuals caught and the species richness for the upstream (US), midstream (MS) and downstream (DS) positions for each of the three barrier status levels assessed in this study.

Barrier Status and Position	Total Number of Caught Individuals (N) per Species (Short Code)										Grand Total	Species Richness
	inga	cmbu	crab	gamb	gldf	grml	leel	seel	srmp	ylml		
Control												
US	661	111	9	105	7	0	0	22	0	0	915	6
MS	9	0	1	0	0	0	0	0	0	0	10	2
DS	905	66	22	58	3	8	2	15	422	23	1524	10
Barrier												
US	0	21	0	1205	38	0	2	64	866	0	2196	6
DS	275	26	12	113	0	14	3	26	357	90	916	9
Non-Barrier												
US	110	16	2	20	28	0	1	0	70	25	272	8
MS	286	15	0	383	6	0	0	4	0	2	696	6
DS	27	42	26	12	0	5	0	2	127	29	270	8

Table 4.3. The total number of caught individuals (N) per species, including the focal inanga, the grand total number of individuals caught and the species richness for the upstream (US), midstream (MS) and downstream (DS) positions for each of the six treatment levels assessed in this study.

Treatment and Position	Total Number of Caught Individuals (N) per Species (Short Code)										Grand Total	Species Richness
	inga	cmbu	crab	gamb	gldf	grml	leel	seel	srmp	ylml		
Control												
US	661	111	9	105	7	0	0	22	0	0	915	6
MS	9	0	1	0	0	0	0	0	0	0	10	2
DS	905	66	22	58	3	8	2	15	422	23	1524	10
Bridge Apron												
US	1	13	0	0	0	0	1	0	0	0	15	3
DS	5	18	0	0	0	0	0	2	0	0	25	3
Culvert												
US	109	3	2	20	0	0	0	0	70	25	229	6
DS	9	24	11	7	0	0	0	0	16	27	94	6
Flood Gate												
US	0	0	0	22	0	0	1	1	824	0	848	4
DS	28	19	8	7	0	2	0	0	204	27	295	7
Pumping Station												
US	0	21	0	1183	38	0	1	63	42	0	1348	6
DS	247	7	4	106	0	12	3	26	153	63	621	9
PS/ FFFG												
US	0	0	0	0	28	0	0	0	0	0	28	1
MS	286	15	0	383	6	0	0	4	0	2	696	6
DS	13	0	15	5	0	5	0	0	111	2	151	6

*The Tutaekuri Waimate stream (twst) was not sampled in this study due to unsuitable fyke net setting conditions. No data is available to assess the passability of the flood gate structure present at this site, though written habitat descriptions and water chemistries are available in Appendix B.

4.2. Inanga and All Species Total Abundances

4.2.1. Inanga Abundance Vs Barrier Status

H_0: Inanga abundance will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'barrier status'.

H_1: Inanga abundance will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'barrier status'.

After controlling for 'position', 'barrier status' and their interaction terms with the downstream (DS) and the control groups, the abundance of inanga in upstream sub-sites above a barrier is not significantly higher than the abundance of inanga in the sub-sites below a barrier or in sites where no structure is present (Fig.4.1). An IRR value of 0 for US barrier sub-sites (95% CI: 0 – Inf., p = 0.9) is not significantly higher than an IRR value of 32.32 for DS barrier sub-sites (95% CI: 15.73 – 66.40, p < 2E-16), failing to reject the null hypothesis that inanga abundance is not higher upstream than it is downstream if migration routes are restricted (Table 4.4) and that drainage structures with a barrier status impede upstream inanga migrations.

Large standard error and wide confidence intervals associated with small sample sizes were expected for the US barrier level as zero inanga individuals were found along these sub-sites.

The abundance of inanga in US and in midstream (MS) sub-sites above a structure categorised as a 'non-barrier' are significantly higher than the abundance of inanga in the sub-sites below a non-barrier structure or in sites where no structure is present (Fig.4.1). The IRR for US non-barrier sub-sites is 5.58 (95% CI: 1.04 – 29.86, p = 0.04) times greater than the IRR of the DS non-barrier sub-sites, and the IRR for MS non-barrier sub-sites is 684.71 (95% CI: 51.46 – 9111.15, p = 7.65 E-07) times greater than the IRR of the DS non-barrier sub-sites, showing that drainage structures with a non-barrier status allow upstream passage for migrating inanga (Table 4.4).

High IRR and wide confidence intervals arose for the MS non-barrier level, as only one sub-site was sampled across the study, and within this sub-site, a substantial number of inanga were found, leading to a single large mean inanga abundance for the entire level, though the standard error remains low compared with the coefficient estimate (Table 4.4).
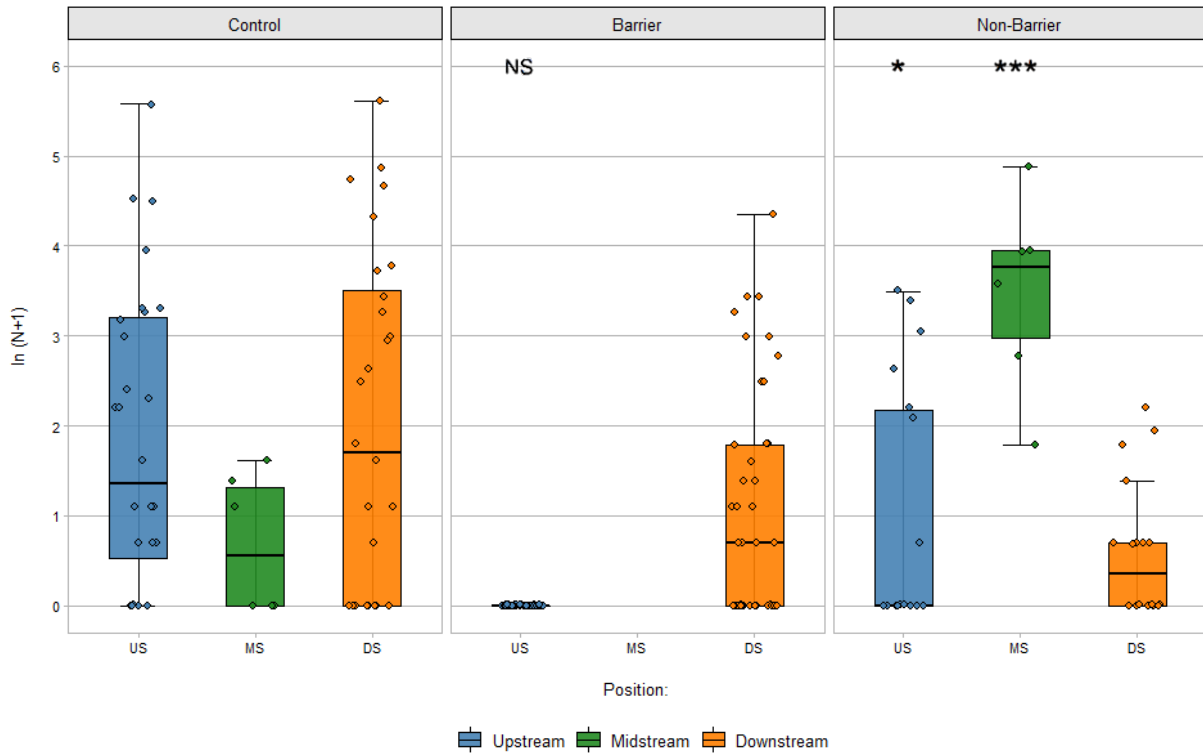
Figure 4.1. Box and whisker and jitter plots of ln(N+1) transformed inanga abundance (N) against upstream (US), midstream (MS) and downstream (DS) positions, by barrier status. Control was the reference group for barrier status and DS was the reference group for the position independent variables. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM).

Table 4.4. Estimated coefficients, their significance (and standard errors) of a negative binomial GLM using the 'Inanga Abundance Vs Barrier Status' dataset. DS was the reference group for the position independent variable and control was the reference for barrier status. SE = Standard Error. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Covariate Levels	Count Model	
	Coefficient	*SE*
(Intercept)	3.48***	(0.37)
MS	-3.07***	(0.93)
US	-0.31	(0.52)
Barrier	-1.55**	(0.48)
Non-Barrier	-3.07***	(0.62)
US: Barrier	-21.92	(2427.23)
MS: Non-Barrier	6.53***	(1.32)
US: Non-Barrier	1.72*	(0.86)

4.2.2. Inanga Abundance Vs Treatment

H$_0$: Inanga abundance will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'treatment'.

H$_1$: Inanga abundance will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'treatment'.

After controlling for 'position', 'treatment' and their interaction terms with the DS and the control groups, the US inanga abundances in sub-sites above a bridge apron, a flood gate, a pumping station or a FFFG drainage structure are not significantly higher than the inanga abundances in the downstream sub-sites below these structures or within sub-sites where no structure is present (Fig.4.2).

The IRR value for US sub-sites above a bridge apron is 0.27 (95% CI: 0.01 – 6.00, p = 0.41) times the IRR value of the DS sub-sites below a bridge apron, showing that the structure does prevent full upstream inanga passage, though a small population of individuals are still able to migrate beyond the structure. However, the IRR values for US sub-sites above a flood gate, a pumping station and a FFFG are each 0 times (95% CI: 0 – Inf., p = 1.00) the DS IRR value of 33.32 (95% CI: 16.68 – 62.64, p < 2E-16), showing that inanga abundances are higher in DS sub-sites below each of these structures and that each of these structures act as full and complete barriers to upstream migrating inanga (Table 4.5).

Large standard errors and wide CIs were expected for levels with a small sample size; in this instance, the total inanga abundance equalled zero for US sub-sites above flood gates, pumping stations and "PS/ FFFG", explaining the high SE and wide CI associated with each covariate level (Table 4.5).

In contrast, after controlling for 'position', 'treatment' and their interaction terms with the DS and the control groups, the US inanga abundance in the sub-site above a culvert and the MS inanga abundance along the "PS/ FFFG" waterway are significantly higher than the inanga abundances in the downstream sub-sites situated below these structures or within sub-sites where no structure is present (Fig.4.2).

The IRR value for the US sub-site above a culvert is 16.58 (95% CI: 1.63 – 168.83, p = 0.02) times greater than the IRR value for the DS sub-site below the culvert, showing that the inanga abundance is considerably greater above the structure than below, and that the inanga migration routes are unrestricted. The IRR value for the MS sub-site above the FFFG along the "PS/ FFFG" waterway is 474.04 (95% CI: 31.14 – 6991.74, p = 7.22 E-06) times greater than the IRR value for the DS sub-site situated below the FFFG, showing that the FFFG structure allows full passage for migrating inanga into upstream habitats and therefore does not act as a migratory barrier (Table 4.5).
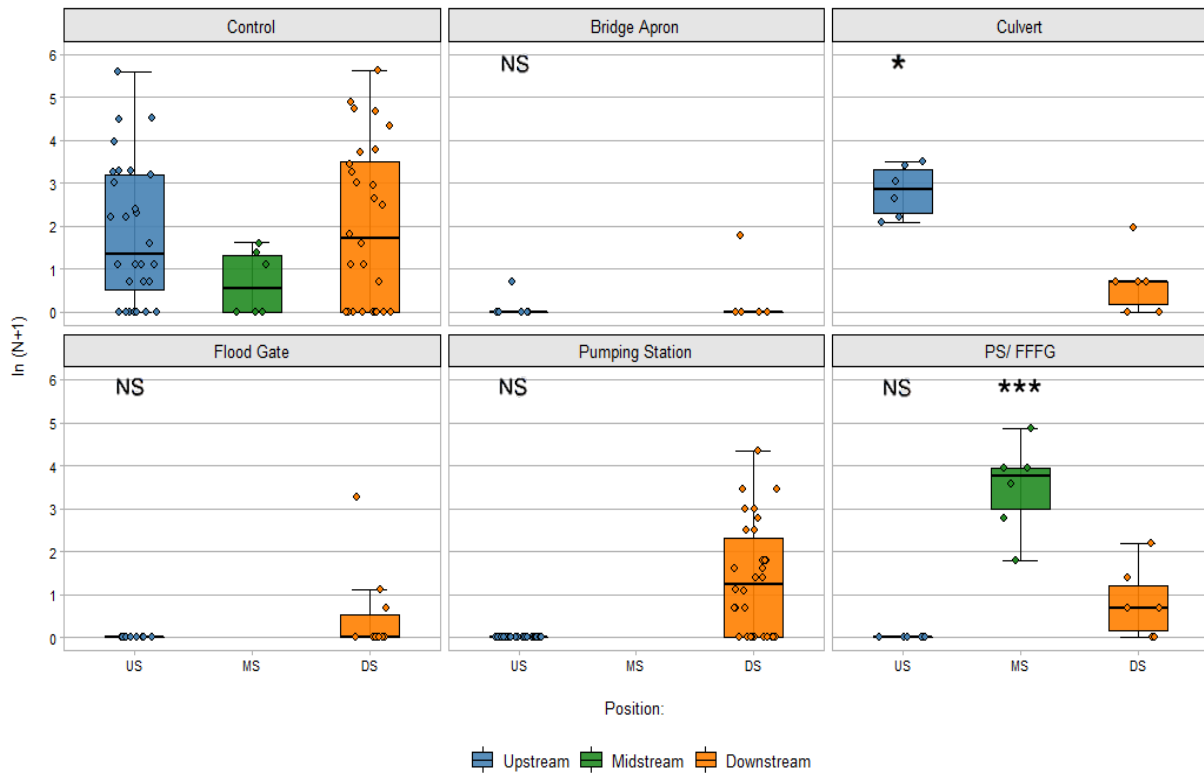
Figure 4.2. Box and whisker and jitter plots of ln(N+1) transformed inanga abundance (N) against upstream (US), midstream (MS) and downstream (DS) positions, by treatment. Control was the reference group for treatment and DS was the reference group for the position independent variables. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM). PS = Pumping Station, FFFG = Fish-Friendly Flood Gate.

Table 4.5. Estimated coefficients, their significance (and standard errors) of a negative binomial GLM using the 'Inanga Abundance Vs Treatment' dataset. DS was the reference group for the position independent variable and control was the reference for treatment. SE = Standard Error. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Covariate Levels	Count Model	
	Coefficient	*SE*
(Intercept)	3.48***	(0.34)
MS	-3.07***	(0.87)
US	-0.31	(0.48)
Bridge Apron	-3.66***	(0.92)
Culvert	-3.07***	(0.87)
Flood Gate	-2.45***	(0.68)
Pumping Station	-1.37**	(0.47)
PS/ FFFG	-2.70**	(0.85)
US: Bridge Apron	-1.30	(1.58)
US: Culvert	2.81*	(1.18)
US: Flood Gate	-21.02	(4486.55)
US: Pumping Station	-22.10	(2886.05)
MS: PS/ FFFG	6.16***	(1.37)
US: PS/ FFFG	-20.76	(6344.94)

4.2.3. All Species Total Abundance Vs Barrier Status

H_0: *The total abundance of all species will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'barrier status'.*

H_1: *The total abundance of all species will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'barrier status'.*

The independent variables 'position', 'barrier status' and their interaction terms were controlled with the downstream and the control reference groups in the negative binomial count component and the zero-inflated logistic component of a zero-inflated negative binomial model.

The total abundances of all species within US sub-sites situated above a barrier and within MS sub-sites situated above a non-barrier are significantly higher than the total abundances of all species in the DS sub-sites below these structures or than the abundances within sub-sites where no structure is present (Fig.4.3). The IRR value for US sub-sites above a barrier is 6.09 (95% CI: 2.41 – 15.35, p = 1.29 E-04) times greater than the IRR value of 6.38 (95% CI: 4.01 – 10.14, p = 4.70 E-15) for DS sub-sites situated below a barrier and the IRR value for MS sub-sites above a non-barrier structure is 132.72 (95% CI: 15.80 – 1114.71, p = 6.73 E-06) times greater than the IRR of the DS sub-sites below a non-barrier. These findings show that the total abundance of all present species, regardless of their ecologies, is greater within US sub-sites above barrier and non-barrier structures than within DS sub-sites below these structures (Table 4.6), despite the barrier structures restricting the upstream migrations of inanga as discovered above. Further analyses of the species composition of each sub-site's fish population has been conducted in the 'Species Diversity and Similarity' section later in this study.

The total abundance of all species within US sub-sites above a non-barrier structure is marginally higher than the total abundance of all species within the DS sub-sites below a non-barrier or within sub-sites where no structure is present (Fig.4.3), although the difference is not significant. The IRR value for US sub-sites above a non-barrier is 1.68 (95% CI: 0.55 – 5.13, p = 0.36) times greater than the IRR value of the DS sub-sites below a non-barrier; an increased abundance which is considerably lower than that found within the US barrier and the MS non-barrier sub-sites (Table 4.6).

Although non-significant, the estimated OR of observing an excess zero in the data for the US sub-sites above a barrier and the US sub-sites above a non-barrier is 42540 times (p = 0.94) and 3.01 times (p = 0.79), respectively, the odds of there being an excess zero in the DS sub-sites and the control data (OR = 0.17, p = 0.05) as key migrating species like inanga were not present within US barrier sub-sites (N = 0), so the probability of obtaining a zero was considerably higher. The estimated odds of obtaining an excess zero in the MS non-barrier sub-site data is lower than the odds of there being an excess zero in the DS sub-site or in the control data (OR = 0.12, p = 0.74) (Table 4.6).
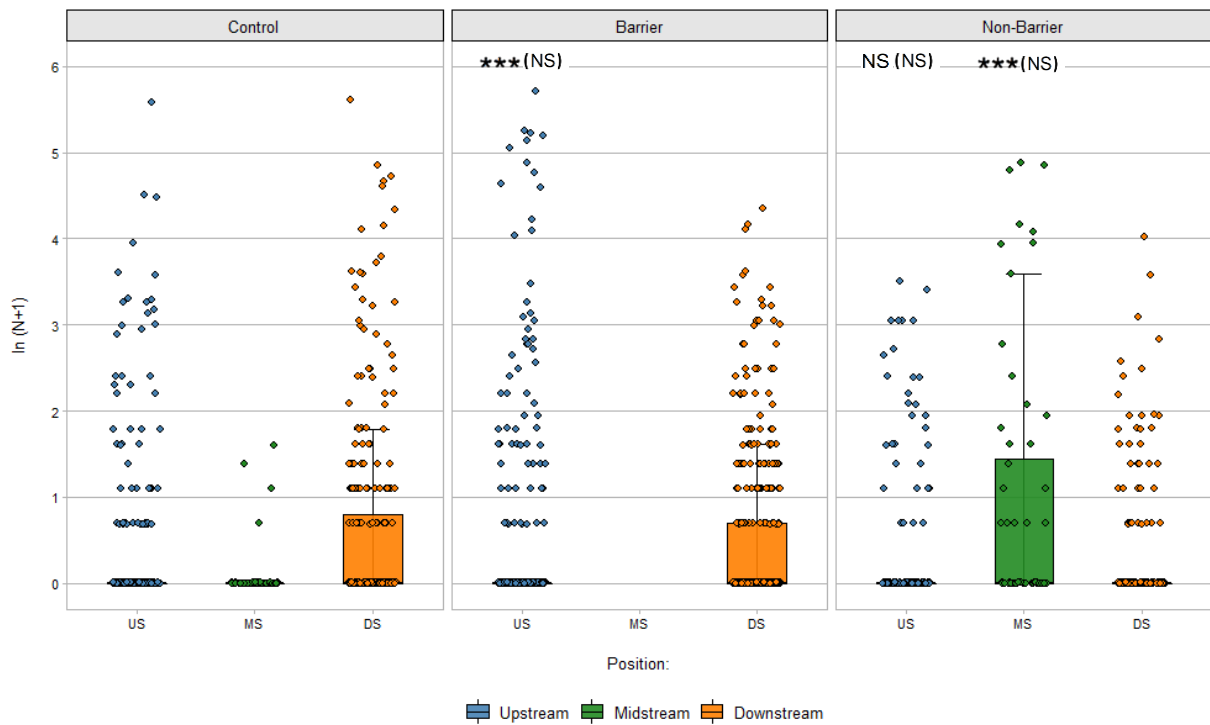
Figure 4.3. Box and whisker and jitter plots of ln(N+1) transformed all species total abundance against upstream (US), midstream (MS) and downstream (DS) positions, by barrier status. Control was the reference group for barrier status and DS was the reference group for the position independent variables. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (zero-inflated negative binomial GLM). Significance levels without brackets indicate count model p-values, significance levels within brackets indicate zero-inflated component p-values.

Table 4.6. Estimated coefficients, their significance (and standard errors) for the negative binomial count component and the zero-inflated logistic component of a zero-inflated negative binomial GLM using the 'All Species Total Abundance Vs Barrier Status' dataset. DS was the reference group for the position independent variable and control was the reference for barrier status. SE = Standard Error. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Covariate Levels	Count Component		Zero-Inflation Component	
	Coefficient	*SE*	Coefficient	*SE*
(Intercept)	1.85***	(0.24)	-1.76*	(0.88)
MS	-2.85**	(0.95)	1.96	(1.43)
US	-0.07	(0.35)	1.56	(0.83)
Barrier	-1.02***	(0.28)	-10.10	(147.10)
Non-Barrier	-1.41***	(0.36)	-1.38	(4.13)
US: Barrier	1.81***	(0.47)	10.66	(147.10)
MS: Non-Barrier	4.89***	(1.09)	-2.11	(6.41)
US: Non-Barrier	0.52	(0.57)	1.10	(4.15)

4.2.4. All Species Total Abundance Vs Treatment

H_0: *The total abundance of all species will not be higher upstream (nor midstream) than it is downstream when upstream migrations are restricted along waterways with varying 'treatment'.*

H_1: *The total abundance of all species will be higher upstream (and midstream) than it is downstream when upstream migrations are unrestricted along waterways with varying 'treatment'.*

'Position', 'structure' and their interaction terms were controlled with the downstream and control reference groups in the negative binomial count component and the zero-inflated logistic component of a zero-inflated negative binomial model.

The total abundances of all species within US sub-sites above a flood gate or a pumping station and the total abundance within the MS sub-site above a FFFG are significantly higher than the total abundance of all species in the DS sub-sites below these structures or within sub-sites where no structure is present (Fig.4.4). The IRR value for US sub-sites above a flood gate is 7.19 (95% CI: 1.60 – 32.34, p = 0.01) times greater than the IRR value of 7.51 (95% CI: 4.82 – 11.68, p < 2E-16) for DS sub-sites situated below a flood gate. The IRR for US sub-sites above a pumping station is 5.80 (95% CI: 2.30 – 14.64, p = 2.01 E-04) times greater than the IRR value for DS sub-sites beneath a pumping station. The IRR value for the MS sub-site above a FFFG along the "PS/FFFG" waterway is 76.84 (95% CI: 8.52 – 692.79, p = 1.09 E-04) times greater than the IRR value for the DS sub-site below the FFFG. The significantly higher abundances found within the US sub-sites above a flood gate or a pumping station and the MS sub-site above a FFFG shows that passageways are unrestricted for some upstream migrating species (Table 4.7), though as the flood gate and the pumping station structures acted as barriers to migrating inanga as previously discovered, further assessments regarding the species composition of each sub-site were required and can be found in the 'Species Diversity and Similarity' section later in this study.

The total abundance of all species within the US sub-site above a bridge apron and within the US "PS/ FFFG" sub-site are not significantly higher than the total abundance of all species within the DS sub-sites situated below these structures or within sub-sites where a structure is not present (Fig.4.4). The IRR value of the US sub-site above a bridge apron is 0.25 (95% CI: 0.02 – 2.80, p = 0.26) times the IRR of the DS bridge apron sub-site and the IRR for the US "PS/FFFG" sub-site is 0.78 (95% CI: 0.09 – 6.61, p = 0.82) times the IRR of the DS sub-site situated below the FFFG along the "PS/FFFG" waterway, showing that the two drainage structures restrict the upstream passageway for some of the waterway's fish population (Table 4.7). The total abundance of all species within the US sub-site above a culvert was higher than the total all species abundance within the DS sub-site situated below the structure, although the difference was not significant (Fig.4.4). The IRR value of the US sub-site is 2.61 (95% CI: 0.78 – 8.75, p = 0.12) times greater than the IRR value of the DS sub-site below the culvert, though the different was lower than observed within the sub-sites where a flood gate or a pumping station was present (Table 4.7).

Although non-significant, the estimated OR of observing an excess zero in the data for the US sub-site above a culvert is 3.40 E+06 times (p = 1.00), a flood gate is 1.94 times (p = 0.39), a pumping station is 1.58 E+05 times (p = 0.99) and the "PS/ FFFG" waterway is 5.60 times (p = 0.26) the odds of there being an excess zero in the DS sub-sites below these structures or within control sub-sites where no structure is present (OR = 0.38, p = 0.03). The estimated odds of obtaining an excess zero in the data for the US bridge apron sub-site (OR = 0.07, p = 0.07) and in the data for the MS "PS/FFFG" sub-site (OR = 0.21, p = 0.43) are lower than in the data for the DS and the control sub-sites (Table 4.7).
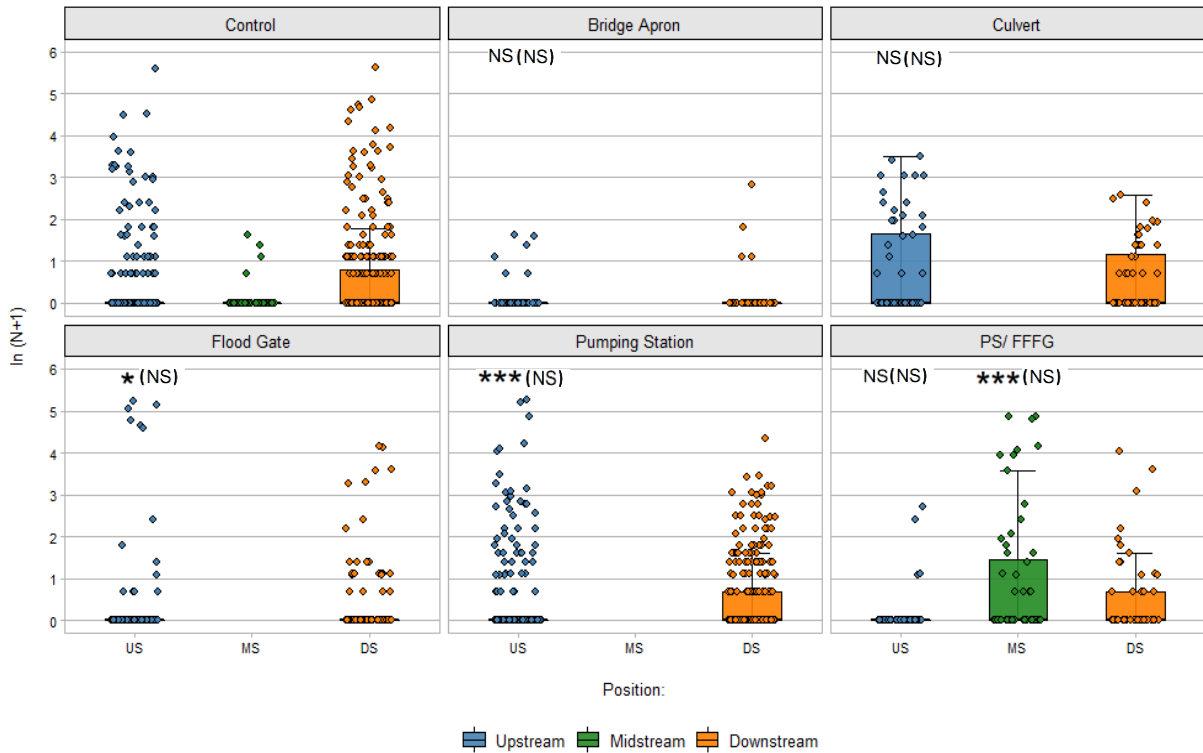
Figure 4.4. Box and whisker and jitter plots of ln(N+1) transformed all species total abundance against upstream (US), midstream (MS) and downstream (DS) positions, by treatment. Control was the reference group for treatment and DS was the reference group for the position independent variables. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (zero-inflated negative binomial GLM). Significance levels without brackets indicate count model p-values, significance levels within brackets indicate zero-inflated component p-values. PS = Pumping Station, FFFG = Fish-Friendly Flood Gate.

Table 4.7. Estimated coefficients, their significance (and standard errors) for the negative binomial count component and the zero-inflated logistic component of a zero-inflated negative binomial GLM using the 'All Species Total Abundance Vs Treatment' dataset. DS was the reference group for the position independent variable and control was the reference for treatment. SE = Standard Error. Significance levels: *** represents p-values < 0.001, ** < 0.01, * <

Covariate Levels	Count Component		Zero-Inflation Component	
	Coefficient	*SE*	Coefficient	*SE*
(Intercept)	2.02***	(0.23)	-0.97*	(0.45)
MS	-2.82**	(0.93)	1.49	(1.10)
US	-0.07	(0.34)	1.10*	(0.45)
Bridge Apron	-1.37	(0.95)	2.25**	(0.81)
Culvert	-1.57***	(0.43)	-27.66	(5.84E+05)
Flood Gate	-0.47	(0.45)	0.44	(0.64)
Pumping Station	-1.29***	(0.28)	-11.77	(1900)
PS/ FFFG	-0.89	(0.51)	-0.52	(1.33)
US: Bridge Apron	-1.39	(1.23)	-2.63	(1.47)
US: Culvert	0.96	(0.62)	15.04	(5.84E+05)

US: Flood Gate	1.97*	(0.77)	0.66	(0.76)
US: Pumping Station	1.76***	(0.47)	11.97	(1900)
MS: PS/ FFFG	4.34***	(1.12)	-1.54	(1.96)
US: PS/ FFFG	-0.25	(1.09)	1.72	(1.52)

4.2.5. Summary of Abundance Results

Table 4.8. Summary of the hypotheses, significance of results and the context at which the results can be applied in situ for the 'Inanga Abundance Vs Barrier Status', 'Inanga Abundance Vs Treatment', 'All Species Total Abundance Vs Barrier Status' and the 'All Species Total Abundance Vs Treatment' datasets. US = Upstream, MS = Midstream, DS = Downstream.

Inanga Abundance Vs Barrier Status

Hypotheses	Levels	Significance	Context
H₀: Inanga abundance will not be higher US (nor MS) than it is DS when upstream migrations are restricted along waterways with varying 'barrier status'.	Barrier	US: Not Sig	Upstream inanga migrations **restricted** by 'barrier' structures
H₁: Inanga abundance will be higher US (and MS) than it is DS when upstream migrations are unrestricted along waterways with varying 'barrier status'.	Non-Barrier	US: Sig MS: Sig	Upstream inanga migrations **NOT restricted** by 'non-barrier' structures

Inanga Abundance Vs Treatment

Hypotheses	Levels	Significance	Context
H₀: Inanga abundance will not be higher US (nor MS) than it is DS when upstream migrations are restricted along waterways with varying 'treatment'.	Bridge Apron	US: Not Sig	Upstream Inanga migrations **restricted**
	Culvert	US: Sig	Upstream Inanga migrations **NOT restricted**
H₁: Inanga abundance will be higher US (and MS) than it is DS when upstream migrations are unrestricted along waterways with varying 'treatment'.	Flood Gate	US: Not Sig	Upstream Inanga migrations **restricted**
	Pumping Station	US: Not Sig	Upstream Inanga migrations **restricted**
	PS/FFFG	US: Not Sig MS: Sig	US: Upstream Inanga migrations **restricted** MS: Upstream Inanga migrations **NOT restricted**

All Species Total Abundance Vs Barrier Status

Hypotheses	Levels	Significance	Context*
H_0: The total abundance of all species will not be higher US (nor MS) than it is DS when upstream migrations are restricted along waterways with varying 'barrier status'.	Barrier	US: Sig	Total abundance of all species **higher** above 'barrier' structures
H_1: The total abundance of all species will be higher US (and MS) than it is DS when upstream migrations are unrestricted along waterways with varying 'barrier status'.	Non-Barrier	US: Not Sig MS: Sig	US: Total abundance of all species **NOT higher** above 'non-barrier' structures MS: Total abundance of all species **higher** above 'non-barrier' structures

All Species Total Abundance Vs Treatment

Hypotheses	Levels	Significance	Context*
H_0: The total abundance of all species will not be higher US (nor MS) than it is DS when upstream migrations are restricted along waterways with varying 'treatment'.	Bridge Apron	US: Not Sig	Total abundance of all species **NOT higher** above a bridge apron
	Culvert	US: Not Sig	Total abundance of all species **NOT higher** above a culvert
H_1: The total abundance of all species will be higher US (and MS) than it is DS when upstream migrations are unrestricted along waterways with varying 'treatment'.	Flood Gate	US: Sig	Total abundance of all species **higher** above a flood gate
	Pumping Station (PS)	US: Sig	Total abundance of all species **higher** above a pumping station
	PS/FFFG	US: Not Sig MS: Sig	US: Total abundance of all species **NOT higher** above a PS MS: Total abundance of all species **higher** above a FFFG

*Although proven to act as a barrier to upstream inanga migrations, the total abundance of all species within some US sub-sites above barrier structures, including flood gates, pumping stations and "PS/ FFFG" is greater than the total abundance of all species within DS sub-sites below these structures. As prementioned, further analyses on the species composition of each sub-site has been conducted later in this study, examining whether these high abundances are solely due to high local populations of invasive or pest species and whether the above structures still act as full barriers to native upstream migrating fish.

4.3. Species Diversity and Similarity

4.3.1. Biodiversity, Richness and Similarity Vs Barrier Status

H_0: Biodiversity, species richness and species evenness upstream (and midstream) will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted along waterways with varying 'barrier status'.

H_1: Biodiversity, species richness and species evenness upstream (and midstream) will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted along waterways with varying 'barrier status'.

The Shannon Weiner diversity index (H') and the species richness for US and MS control sub-sites are significantly different to the H' index and the species richness of the DS control sub-sites, rejecting the null hypothesis in favour of the alternative that the biodiversity between the US and the DS sub-sites, and the MS and the DS sub-sites differs if upstream fish passage is restricted (Fig.4.5). An US ENS value of 2.49 (H'_{max} = 1.79, p = 1.19 E-07) and a MS ENS value of 1.39 (H'_{max} = 0.69, p = 0.004) are lower than the DS ENS value of 3.14 (H'_{max} = 2.30), showing that a greater number of species inhabit the lower reaches of the waterways, which is generally expected for sub-sites closer to the freshwater-saltwater boundary zone. However, Pielou's evenness index (J') was similar between the US and DS sub-sites and the MS and DS sub-sites at values of around 0.5, indicating that the abundances of individuals relative to the species richness did not differ between sub-sites.

The biodiversity values and species richness of the US sub-sites above a barrier are significantly different to the biodiversity values and species richness of the DS sub-sites below a barrier. An US ENS value of 2.51 (H'_{max} = 1.79, p = 9.12 E-73) is lower than a DS ENS value of 4.74 (H'_{max} = 2.20), showing that a greater number of species inhabit the DS sub-sites than the US and that structures categorised as 'barriers' restrict the upstream movement of numerous fish species. A lower J' value for the US sub-site compared with the J' value for the DS sub-site also indicates that barrier structures lead to considerable differences in the abundance of individuals relative to species richness.

The H' values and the species richness of the US sub-sites above non-barrier structures are not significantly different to the H' values and the species richness of the DS sub-sites below non-barrier structures. An US ENS value of 4.88 (H'_{max} = 2.08, p = 0.96) is very similar to a DS ENS value of 4.89 (H'_{max} = 2.08), showing that an equal number of species are present both above and below a non-barrier and that these structures do not restrict the upstream movement of fish, which similar J' values between sub-sites confirms. In contrast, the H' values for the MS and the DS sub-sites are significantly different as a MS ENS value of 2.37 (H'_{max} = 1.79, p = 3.09 E-28) is considerably lower than the ENS value of the DS sub-site, though this may be due to the fact that only one MS non-barrier sub-site was sampled across this study and that this may have naturally contributed to a lower total number of caught species (Fig.4.5).
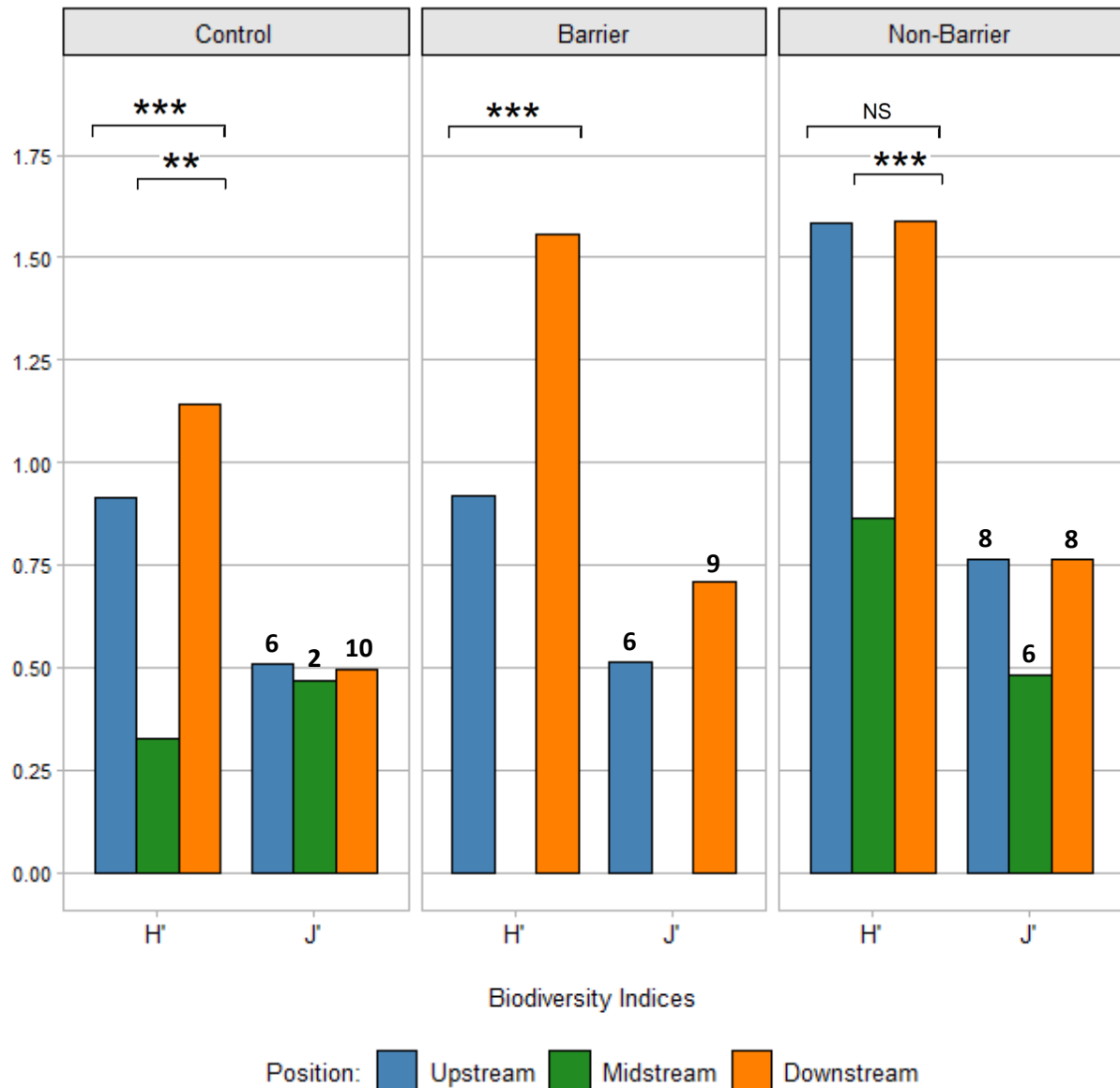
Figure 4.5. Shannon Weiner diversity index (H') and Pielou's evenness index (J') against upstream (US), midstream (MS) and downstream (DS) positions, by barrier status. Numbers in **bold** represent the species richness of US, MS and DS positions, by barrier status. Hutcheson t-test comparing the statistical significance of the difference in H' between US and DS, and MS and DS sub-sites of each barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05.

The Sørensen's similarity index (ß) was calculated for every possible sub-site position pair along waterways with differing 'Barrier Status'. A species present along both sub-sites within a position pair was noted as a shared species (Table 4.9).

ß was greatest between the US and the DS control sub-sites (75% similarity with 6 shared species), though ß was the lowest between the US and the MS control sub-sites (50% similarity with 2 shared species) and the MS and the DS control sub-sites (33% similarity with 2 shared species). Migratory inanga were present across all three sub-sites however, indicating that migration routes were unrestricted across the control waterways and that inanga inhabit each of the sub-sites.

ß was also highest between the US and the DS sub-sites (75% with 6 shared species), the US and the MS sub-sites (71% with 5 shared species) and the MS and the DS sub-sites (71% with 5 shared species) where a non-barrier structure was present showing that a large number of species, including inanga and yellow-eye mullet, were able to migrate beyond structures categorised as non-barriers into upstream sub-sites and that these structures did not restrict the upstream migrations of specialist native fish.

The US and the DS sub-sites where a barrier was present had a species similarity of 67% and shared 5 species, though specialist native migrating fish like inanga and yellow-eye mullet were absent from the US sub-site and were only found DS, indicating that barrier structures fully restrict the upstream migrations of these ecologically important species, though allow invasive or more environmentally tolerant species like common bully and gambusia to inhabit.

Table 4.9. Sørensen's similarity index (ß) and the shared species short names for upstream (US), midstream (MS) and downstream (DS) sub-site position pairs, by barrier status.

Barrier Status	Position Pairs	ß	Shared Species (Short Code)
barrier	US-DS	0.67	cmbu, gamb, leel, seel, srmp
control	US-MS	0.50	inga, crab
control	MS-DS	0.33	inga, crab
control	US-DS	0.75	inga, cmbu, crab, gamb, gldf, seel
non-barrier	US-MS	0.71	inga, cmbu, gamb, gldf, ylml
non-barrier	MS-DS	0.71	inga, cmbu, gamb, seel, ylml
non-barrier	US-DS	0.75	inga, cmbu, crab, gamb, srmp, ylml

4.3.2. Biodiversity, Richness and Similarity Vs Treatment

H_0: *Biodiversity, species richness and species evenness upstream (and midstream) will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted along waterways with varying 'treatment'.*

H_1: *Biodiversity, species richness and species evenness upstream (and midstream) will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted along waterways with varying 'treatment'.*

The Shannon Weiner diversity index (H') for the US sub-site above a bridge apron is slightly lower than the H' index for the DS sub-site below a bridge apron, although the difference is not significant (Fig.4.6). The same species richness values for both sub-sites shows that the structure does not act as a barrier to upstream migrating species, though a lower US ENS value of 1.62 (H'_{max} = 1.10, p = 0.33) compared to a DS ENS value of 2.14 (H'_{max} = 1.10) indicates that there is a higher relative abundance of fish within the DS sub-site and that the structure therefore acts as a partial barrier to some individuals, which a slightly higher DS Pielou's evenness index (J') confirms.

The H' index of the US sub-site above a culvert is significantly different to the H' index of the DS sub-site below a culvert (Fig.4.6). An US ENS value of 3.56 (H'_{max} = 1.79, p = 2.82E-08) is lower than the DS ENS value of 5.35 (H'_{max} = 1.79) showing that a greater biodiversity of fish, relative to their abundance was present within the DS sub-site. However, the same level of species richness and similar J' values between both sub-sites indicates that the culvert does not act as a full barrier to upstream migrating species but instead acts as a partial barrier to some individuals.

The biodiversity and the species richness of the US sub-sites above a flood gate and a pumping station are significantly different to the biodiversity and the species richness of the DS sub-sites below a flood gate and a pumping station (Fig.4.6). An US flood gate ENS value of 1.15 (H'_{max} = 1.39, p = 1.37 E-34) is lower than the DS flood gate ENS value of 2.99 (H'_{max} = 1.95), whilst the US pumping station ENS value of 1.71 (H'_{max} = 1.79, p = 8.57 E-100) is considerably lower than the DS pumping station ENS value of 4.78 (H'_{max} = 2.20). These lower US ENS values, along with lower US J' values, indicate that a greater biodiversity of species are present within the DS sub-sites and that the structures act as fully restrictive barriers to numerous upstream migrating fish species.

As discovered above, the H' index and the species richness of the US sub-site above the pumping station (PS) along the "PS/ FFFG" waterway is significantly different to the H' index and the species richness of the DS sub-site below the FFFG (Fig.4.6). An US ENS value of 1 (H'_{max} = 0, p = 1.47 E-19) was considerably lower than the DS ENS value of 2.58 (H'_{max} = 1.79), which, along with drastically different J' values, show that there is a greater number of species present within the DS sub-site and that the pumping station acts as a fully restrictive barrier to upstream migrating fish. In contrast, the H' index and the species richness of the MS sub-site above the FFFG is not significantly different to the H' index and the species richness of the DS sub-site below the FFFG. A similar MS ENS value of 2.37 (H'_{max} = 1.79, p = 0.37) and a DS ENS value of 2.58 (H'_{max} = 1.79), along with similar J' values indicate that both sub-sites exhibit a similar biodiversity of species and that the FFFG does not act as a barrier to upstream migrating fish.

Comparisons of the H' index, species richness and J' values between US, MS and DS control sub-sites are the same as those reported in the above 'Barrier Status' section.
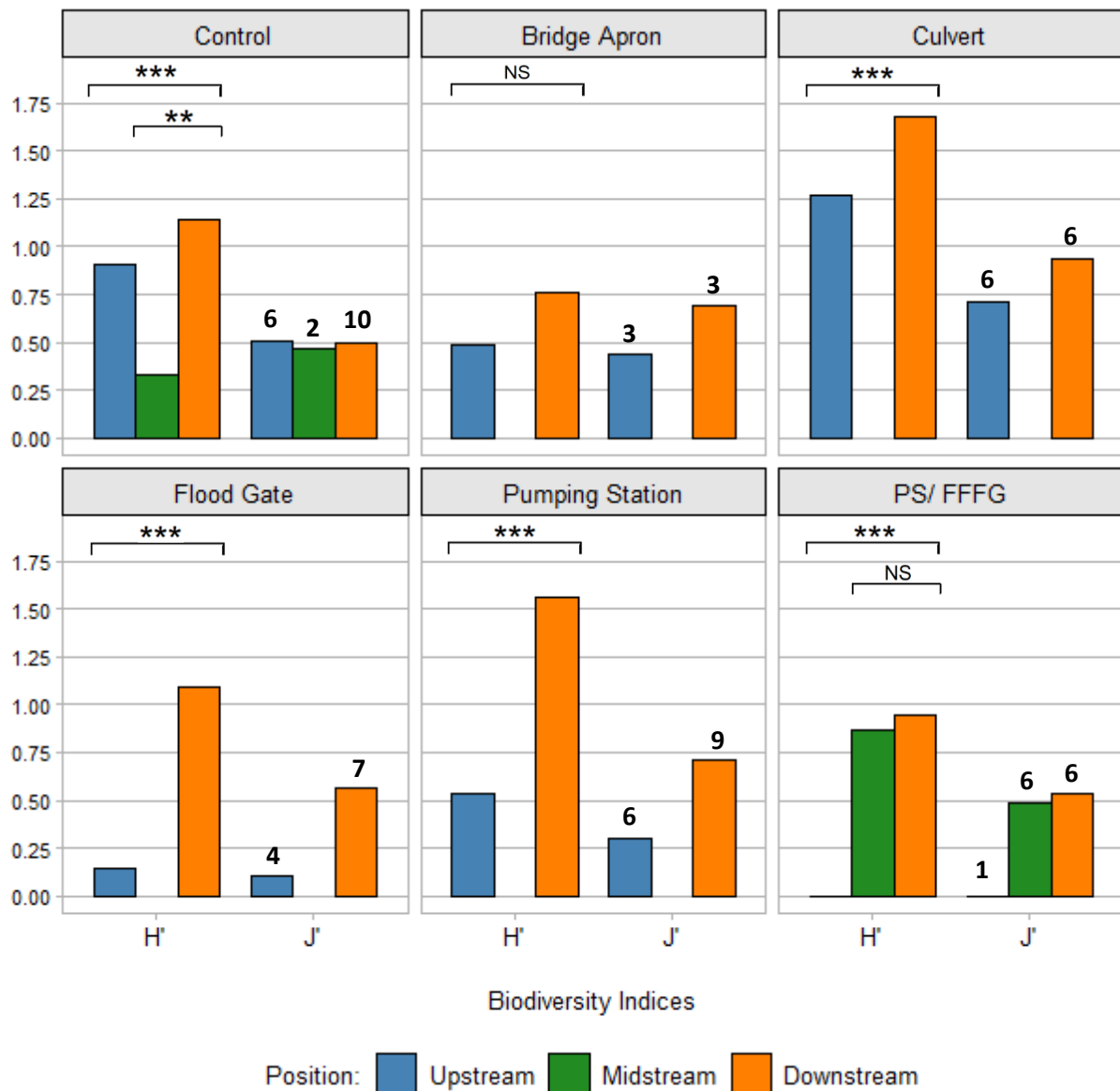
Figure 4.6. Shannon Weiner diversity index (H') and Pielou's evenness index (J') against upstream (US), midstream (MS) and downstream (DS) positions, by treatment. Numbers in **bold** represent the species richness of US, MS and DS positions, by treatment. Hutcheson t-test comparing the statistical significance of the difference in H' between US or MS and DS subsites of each treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05.

As with the 'Barrier Status' dataset, Sørensen's similarity index (ß) was calculated for each possible sub-site position pair along sites with differing 'Treatments'. Shared species represent species which were present along both sub-sites within a position pair (Table 4.10).

ß values for the position pairs of the control sub-sites are the same as those reported in the above 'Barrier Status' section.

ß values are high for the two partial barriers, or non-barrier structures assessed in this study. The highest ß values are between the US and the DS bridge apron position pair (67% similarity with 2 shared species) and the US and the DS culvert position pair (100% similarity with 6 shared species) with the specialist, native inanga being present within all sub-sites, further indicating that neither structure acts as a fully restrictive barrier to upstream migrating fish.

One of the lowest ß values across all position pairs is between the US and the DS sub-sites where a flood gate is present (36% similarity with 2 shared species). Native migrating species like inanga and yellow-eye mullet are absent within the US sub-site, which comprises only of invasive gambusia and the environmentally tolerant New Zealand freshwater shrimp, indicating that the structure fully restricts upstream migrating species but allows the flourishing of upstream pest populations. A higher ß value for the US and the DS pumping station position pair (67% similarity with 5 shared species) is observed, though native migratory species like inanga remain absent from US sub-sites and the structure is still considered as a fully restrictive migratory barrier.

The lowest position pair ß values across all sub-sites are between the US and the DS sub-sites (0% similarity with 0 shared species) and the US and the MS sub-sites (29% similarity with 1 shared species) along the "PS/ FFFG" waterway, indicating that the pumping station (PS) acts as a full restrictive barrier to all upstream migrating species and that the invasive goldfish is the only species present within the US and the MS or the DS sub-sites. A higher ß value for the MS and the DS sub-site position pair along the "PS/ FFFG" waterway (50% similarity with 3 shared species) is observed, where the native migratory inanga and yellow-eye mullet are present within both sub-sites, indicating that the FFFG does not restrict full upstream passage and is therefore not regarded as a migratory barrier.

Table 4.10. Sørensen's similarity index (ß) and the shared species short names for upstream (US), midstream (MS) and downstream (DS) sub-site position pairs, by treatment.

Treatment	Position Pairs	ß	Shared Species (Short Code)
bridge apron	US-DS	0.67	inga, cmbu
control	US-MS	0.50	inga, crab
control	MS-DS	0.33	inga, crab
control	US-DS	0.75	inga, cmbu, crab, gamb, gldf, seel
culvert	US-DS	1	inga, cmbu, crab, gamb, srmp, ylml
flood gate	US-DS	0.36	gamb, srmp
pumping station	US-DS	0.67	cmbu, gamb, leel, seel, srmp
PS/FFFG	US-MS	0.29	gldf
PS/FFFG	MS-DS	0.50	inga, gamb, ylml
PS/FFFG	US-DS	0	-

4.3.3. Summary of Biodiversity, Richness and Similarity Results

Table 4.11. Summary of the hypotheses, significance of results and the context at which the results can be applied in situ for the 'Biodiversity, Richness and Similarity Vs Barrier Status' and the 'Biodiversity, Richness and Similarity Vs Treatment' analyses. US = Upstream, MS = Midstream, DS = Downstream. ß = Sørensen's similarity index. inga = inanga short code.

Biodiversity, Richness and Similarity Vs Barrier Status

Hypotheses	Levels	Significance	Context
H_0: *Biodiversity, species richness and species evenness US (and MS) will not differ to the biodiversity, species richness and species evenness DS if upstream fish passage is unrestricted along waterways with varying 'barrier status'.* H_1: *Biodiversity, species richness and species evenness US (and MS) will differ to the biodiversity, species richness and species evenness DS if upstream fish passage is restricted along waterways with varying 'barrier status'.*	Control	US: Sig MS: Sig ß: inga found US, MS and DS	US and MS: Upstream and midstream biodiversity lower than downstream, though inga present ∴ migrations **NOT restricted** when no structure is present
	Barrier	US: Sig ß: inga absent from US	Upstream biodiversity lower than downstream; inga not present US ∴ migrations **restricted** by 'barrier' structures
	Non-Barrier	US: Not Sig MS: Sig ß: inga found US, MS and DS	US: Upstream biodiversity similar to downstream; inga present ∴ migrations **NOT restricted** by 'non-barrier' structures MS: Midstream biodiversity lower than downstream, though inga present ∴ migrations **NOT restricted** by 'non-barrier' structures

Biodiversity, Richness and Similarity Vs Treatment

Hypotheses	Levels	Significance	Context
H_0: *Biodiversity, species richness and species evenness US (and MS) will not differ to the biodiversity, species richness and species evenness DS if upstream fish passage is unrestricted along waterways with varying 'treatment'.*	Control	Same as above	–
	Bridge Apron	US: Not Sig ß: inga found US and DS	Upstream biodiversity lower than downstream; though inga present ∴ migrations **NOT restricted**
	Culvert	US: Sig	Upstream biodiversity lower than downstream; though inga present

H₁: Biodiversity, species richness and species evenness US (and MS) will differ to the biodiversity, species richness and species evenness DS if upstream fish passage is restricted along waterways with varying 'treatment'.		ß: inga found US and DS	∴ migrations **NOT restricted**
	Flood Gate	US: Sig	Upstream biodiversity lower than downstream; inga not present US
		ß: inga absent from US	∴ migrations **restricted**
	Pumping Station (PS)	US: Sig	Upstream biodiversity lower than downstream; inga not present US
		ß: inga absent from US	∴ migrations **restricted**
	PS/FFFG	US: Sig MS: Not Sig	US: Upstream biodiversity lower than downstream; inga not present US
			∴ migrations **restricted** by PS
		ß: inga found MS and DS; absent from US	
			MS: Midstream biodiversity similar to downstream; inga present ∴ migrations **NOT restricted** by FFFG

4.4. The Effects of Environmental Factors on Species Abundance

4.4.1. Data Description for Environmental Analyses

As with the data used in the abundance and the biodiversity analyses, the inanga and the all species total abundances refers to the total number of caught inanga individuals (N) and the grand total number of caught individuals across all species (N) in each of the upstream, the midstream and the downstream sub-sites along waterways, or "sites" with varying barrier status and treatment (Table 2.1). For each sub-site, the number of caught inanga individuals and the total number of caught individuals of all species were pooled across all set fyke nets to give the total inanga abundance and the grand total of all species abundance as reported in Table 4.12. These abundances were then used as the dependent variables in the following analyses and compared against the mean temperature (°C), the mean dissolved oxygen saturation (%) and the mean specific conductivity (µS/cm) of each sub-site.

As reported in the Methods section 3.2.3., the above environmental measures were collected in the morning and in the afternoon for each sub-site along each of the 16 sampled waterways to account for any varying climatic conditions throughout the day. Daily averages were calculated from the two readings and are reported as means in Table 4.12, noting that unsuitable environmental conditions at the Tutaekuri Waimate stream (twst) prevented fyke nets to be set, so fish abundance data could not be obtained, and the site was therefore excluded from analyses.

Statistical analyses categorised the now 15 sites into the three 'barrier status' levels: control, barrier and non-barrier, and into the six 'treatment' levels: control, bridge apron, culvert, flood gate, PS/ FFFG and pumping station which were used to assess whether the inanga or the total of all species abundances were impacted by varying temperature, dissolved oxygen saturation and specific conductivities in sites with varying barrier status or treatment.

Brief observational comparisons of the mean temperature, the mean dissolved oxygen and the mean specific conductivity between sub-sites along each individual site are included in Appendix B using the data reported in Table 4.12.

4.4.2. Tidal Variation

The three environmental variables assessed in this study: temperature, dissolved oxygen saturation and specific conductivity are all co-correlated, meaning that a change in one may directly influence another, which interaction terms in the statistical analyses have accounted for. However, additional natural factors like tidal variation and associated factors like the time of the day or the height of the tide may also influence a particular environmental variable. For example, a higher tide may be associated with increased specific conductivity and reduced water temperature, in particular within downstream sub-sites and control waterways where no drainage structure is present. The analyses in this study however do not take into account the influence of tidal variation, though Table 4.13 describes the time, the height of the tide and the relative position of the tide along the tidal cycle for the exact day at which fish abundance and environmental measures were taken along each specific waterway. Observational assessments can be made between Tables 4.12 and 4.13, comparing the abundances along each site to the tidal features on the day of fish sampling for that particular site. The effect of tidal variation on each of the three environmental variables will be discussed in greater detail in the Discussion section 5.3.1., though no further assessments on the effect of tidal variation on inanga and all species total abundances are made in this study.

Table 4.12. Inanga and all species total abundances (N) and mean daily water chemistries for the upstream (US), the midstream (MS) and the downstream (DS) sub-sites along the 16 sampled waterways in this study with their respected barrier status and treatment categories. Dash (-) represents abundances and water qualities for the "twst" site which were not measured due to unsuitable environmental conditions and were therefore not included in the analyses. PS = Pumping Station, FFFG = Fish-Friendly Flood Gate, Temp = Temperature, DO = Dissolved Oxygen Saturation, SPC = Specific Conductivity. Refer to Methods section 3.2.3. for US, MS and DS site layouts and for descriptions on how the mean water chemistries were calculated.

Site Short Code	Barrier Status	Treatment	Total Number of Caught Inanga Individuals (N)			Grand Total Number of Caught Individuals of All Species (N)			Mean Temp (°C)			Mean DO (%)			Mean SPC (µS/cm)		
Position			US	MS	DS	US	MS	DS	US	MS	DS	US	MS	DS	US	MS	DS
taup	Control	Control	124		657	135		678	18.6		18.1	90.2		30.9	363		459
talm	Control	Control	28		71	148		201	20.3		23.8	78.6		156.9	382		9100
grcr	Control	Control	462		28	468		249	19.0		19.6	83.8		119.2	1106		22220
tnst	Control	Control	6		0	121		39	20.1		18.0	95.8		79.2	372		434
past	Control	Control	41	9	149	43	10	357	18.6	18.6	21.6	86.5	87.9	97.5	436	436	1164
mgst	Non-Barrier	Bridge Apron	1		5	15		25	16.8		16.9	140.8		135.8	606		596
ngbw	Non-Barrier	Culvert	109		9	229		94	22.0		22.4	78.4		92.8	18106		15260
twst	Barrier	Flood Gate	-		-	-		-	17.9		-	113.7		-	2339		-
paps	Barrier	Flood Gate	0		0	23		46	22.1		22.6	55.4		96.4	689		1621
mpdr	Barrier	Flood Gate	0		28	825		249	17.7		19.6	69.8		119.2	24311		22220
mucr	Barrier	PS/ FFFG	0	286	13	28	696	151	21.5	21.4	21.6	114.9	49.0	104.5	1869	10102	18528
codr	Barrier	Pumping Station	0		102	64		123	25.1		24.9	103.3		142.0	2122		12897
otri	Barrier	Pumping Station	0		0	110		37	23.1		19.7	81.1		143.2	4058		12861
pust	Barrier	Pumping Station	0		71	255		201	21.3		23.8	101.4		156.9	810		9100
asdr	Barrier	Pumping Station	0		71	881		201	20.0		23.8	109.8		156.9	3959		9100
pldr	Barrier	Pumping Station	0		3	38		59	18.4		18.5	50.5		74.2	429		3674

Table 4.13. The time, the height (m) and the relative position on the tide cycle of each of the four tides on the date where fish abundances and environmental measures were taken for each of the 16 lowland waterways in this study. T = Tidal, NT = Non-Tidal, US = Upstream, DS = Downstream. Dash (-) along tidal features for non-tidal sites where environmental measures would not have been influenced by the tide.

Site Short Code	Tidal/ Non-Tidal	Date Sampled	1st Tide			2nd Tide			3rd Tide			4th Tide		
			Time	*Height (m)*	*Position of Tide Cycle*	*Time*	*Height (m)*	*Position of Tide Cycle*	*Time*	*Height (m)*	*Position of Tide Cycle*	*Time*	*Height (m)*	*Position of Tide Cycle*
taup	NT	12/12/2019	-	-	-	-	-	-	-	-	-	-	-	-
talm	T	13/12/2019	00:55	0.3	Low	07:06	1.7	High	13:22	0.3	Low	19:30	1.6	High
grcr	T	08/01/2020	04:07	1.5	High	10:22	0.4	Low	16:28	1.5	High	22:43	0.4	Low
tnst	T	22/01/2020	04:11	1.7	High	10:26	0.3	Low	16:33	1.6	High	22:48	0.3	Low
past	T	23/01/2020	05:07	1.7	High	11:21	0.3	Low	17:27	1.6	High	23:42	0.3	Low
mgst	NT	19/12/2019	-	-	-	-	-	-	-	-	-	-	-	-
ngbw	T	21/01/2020	03:13	1.7	High	09:29	0.3	Low	15:37	1.6	High	21:53	0.3	Low
twst	US = NT DS = T	04/12/2019	00:01	1.5	High	06:09	0.4	Low	12:26	1.6	High	18:45	0.3	Low
paps	T	06/12/2019	01:36	1.5	High	07:46	0.4	Low	14:01	1.6	High	20:18	0.4	Low
mpdr	T	07/01/2020	03:18	1.5	High	09:33	0.5	Low	15:40	1.5	High	21:55	0.4	Low
mucr	T	09/01/2020	04:56	1.6	High	11:12	0.4	Low	17:18	1.6	High	23:32	0.3	Low
codr	T	28/11/2019	01:17	0.1	Low	07:33	1.8	High	13:47	0.1	Low	19:55	1.8	High
otri	T	05/12/2019	00:48	1.5	High	06:56	0.4	Low	13:13	1.6	High	19:32	0.4	Low
pust	T	11/12/2019	05:35	1.6	High	11:48	0.4	Low	17:54	1.5	High	00:09	0.3	Low
asdr	T	20/12/2019	00:38	1.7	High	06:48	0.2	Low	13:04	1.8	High	19:24	0.2	Low
pldr	T	18/12/2019	04:59	0.2	Low	11:14	1.8	High	17:36	0.2	Low	23:43	1.7	High

4.4.3. Inanga Abundance Vs Barrier Status

H_0: Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the abundance of inanga along waterways with varying 'barrier status'.

H_1: Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the abundance of inanga along waterways with varying 'barrier status'.

After scaling the three environmental fixed effects and considering the variance of the nested barrier status and position random effects, the abundance of inanga along waterways with a control, a barrier and a non-barrier status significantly differed as the mean temperature and the mean dissolved oxygen increased, rejecting the null hypothesis in favour of the alternative that temperature and dissolved oxygen do have an effect on the abundance of inanga (Fig.4.7; Fig.4.8).

For every 1 standard deviation (SD) increase in mean temperature, the IRR value of the predicted inanga abundance significantly increases by 2.34 (95% CI: 1.46 – 3.75) compared with an intercept IRR of 3.15 (95% CI: 0.50 – 19.89), whilst holding all other independent variables constant, noting that IRR values greater than 1 correspond to an increase in abundance and values less than 1 correspond to a decrease in abundance (Table 4.14). Observing the EMM plot (Fig.4.7), the predicted mean inanga abundance increased across all upstream, midstream and downstream positions along the control, the barrier and the non-barrier sites, although the greatest increase in predicted abundance was observed between 22-24°C along upstream control sites, downstream barrier sites and midstream non-barrier sites. No inanga inhabited upstream sub-sites above waterways categorised as a barrier, so predicted counts were zero across all temperatures. These findings show that inanga are most abundant in warmer waterways in this study, assuming that passage is unrestricted by a barrier structure.

For every 1 SD increase in mean dissolved oxygen saturation, inanga abundance significantly decreases by an IRR of 0.43 (95% CI: 0.28 – 0.66) compared to the intercept (Table 4.14), showing that inanga abundance is lower in waterways which have a higher dissolved oxygen saturation in this study. In direct contrast to the mean temperature findings, the EMM plot (Fig.4.8) show that the predicted mean inanga abundance decreases as the mean dissolved oxygen saturation increases for all positions across all waterways. The greatest abundances were between 30-50% DO saturation, whilst zero inanga are predicted to inhabit sites where the DO saturation exceeds ~140% (Fig.4.8).

The predicted inanga abundance increases by an IRR of 1.10 (95% CI: 0.64 – 1.89) for every 1 SD increase in mean specific conductivity, although the increase is not significantly different (Table 4.14). The EMM plot shows that the predicted mean inanga abundance increases across all positions along all barrier status types as the SPC increases, although little change is observed within the midstream control sites and the upstream and the downstream non-barrier sites, where an almost equal number of inanga are predicted to inhabit (Fig.4.9) These findings indicate that the abundance of inanga is generally not influenced by a waterway's specific conductivity in this study.

Examining the marginal R^2 value, or the variance explained by the fixed environmental effects alone, it is shown that only 12.67% of the variance associated with the inanga abundance has been accounted for. However, when the variance explained by the random position and barrier status effects is also considered, the total variance, or the conditional R^2 value rises to 98.71%, indicating that the regression model fits the dataset well when both of the fixed and the random effects are included (Table 4.14).
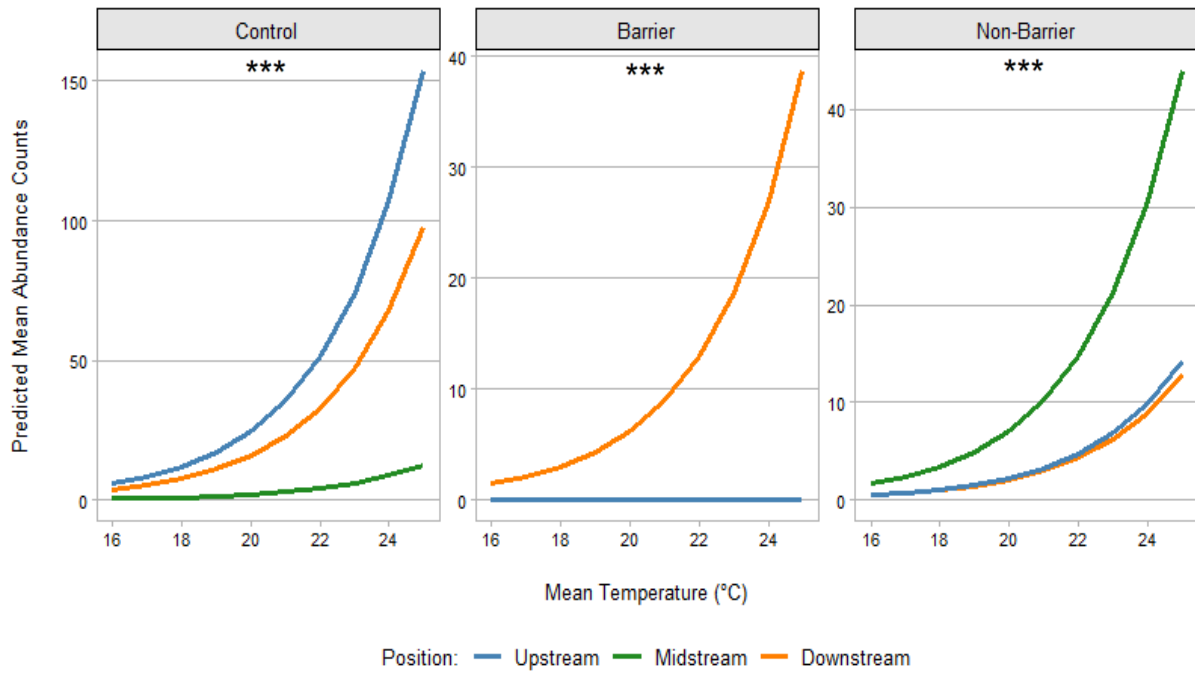
Figure 4.7. Estimated Marginal Mean (EMM) plots of predicted mean inanga abundance against mean temperature (°C) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
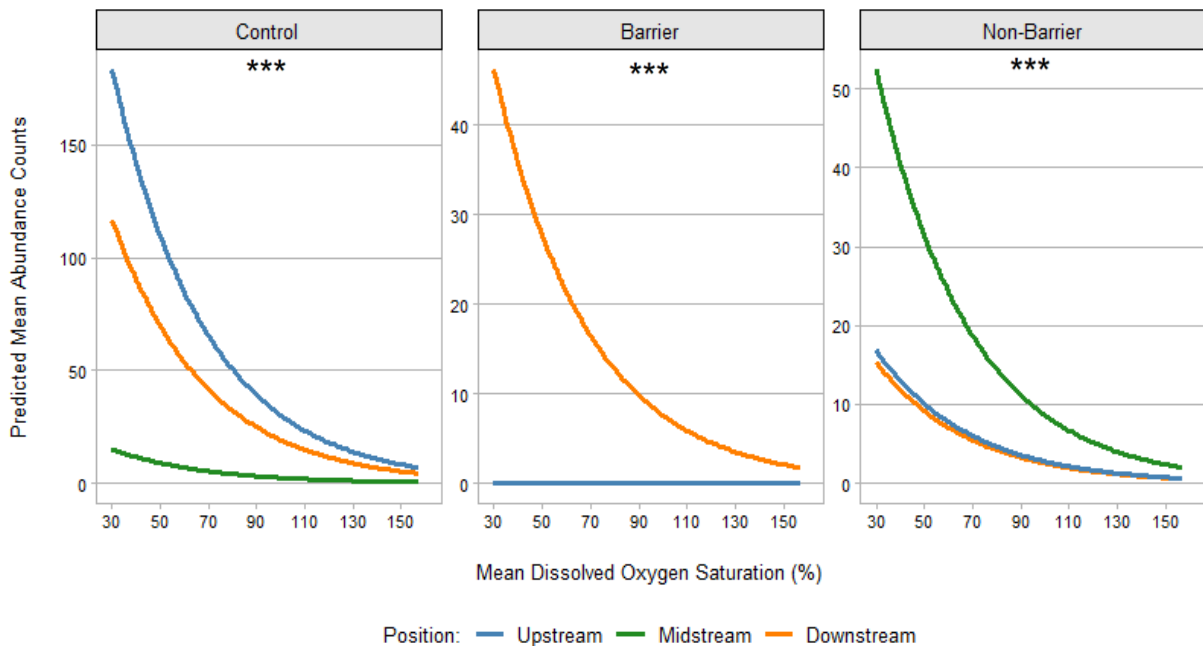
Figure 4.8. Estimated Marginal Mean (EMM) plots of predicted mean inanga abundance counts against mean dissolved oxygen saturation (%) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).

Figure 4.9. Estimated Marginal Mean (EMM) plots of predicted mean inanga abundance against mean specific conductivity (μS/cm) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).

Table 4.14. Estimated coefficients, their significance, (standard errors), z values and p-values for the scaled fixed effects and the variance and standard deviation (SD) for the nested random effects of a negative binomial GLMM using the 'Inanga Abundance Vs Barrier Status' dataset. SE = Standard Error, Temp = Temperature (°C), DO = Dissolved Oxygen Saturation (%), SPC = Specific Conductivity (μS/cm). Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Fixed Effects				
Scaled Covariate	Coefficient	*SE*	z value	p-value
(Intercept)	1.15	(0.94)	1.22	0.22
Mean Temp	0.85***	(0.24)	3.52	4.39E-04
Mean DO	-0.85***	(0.22)	-3.84	1.21E-04
Mean SPC	0.10	(0.28)	0.37	0.72

Random Effects			
Group	Name	Variance	SD
Barrier Status: Position	(Intercept)	6.50	2.55
Barrier Status	(Intercept)	1.12E-06	1.06E-03
Residual		0.73	0.54

Marginal R^2/ Conditional R^2		0.13/ 0.99	

4.4.4. Inanga Abundance Vs Treatment

H_0: *Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the abundance of inanga along waterways with varying 'treatment'.*

H_1: *Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the abundance of inanga along waterways with varying 'treatment'.*

After scaling the fixed environmental effects and considering the nested treatment and position random effects, the abundance of inanga significantly differed as the mean temperature and the mean dissolved oxygen saturation increased across waterways with varying treatment types.

The abundance of inanga significantly increases by an IRR value of 3.38 (95% CI: 1.28 − 8.89) for every 1 SD increase in mean temperature, compared with an intercept IRR of 1.50 (95% CI: 0.28-8.03) (Table 4.15) across all treatment types, although the difference varied considerably depending on the position (Fig.4.10). The mean predicted inanga abundance shown on the EMM plot (Fig.4.10) indicates that the largest increase occurred between 22-24°C along upstream control and culvert sub-sites, the midstream PS/ FFFG sub-site and along downstream sub-sites situated below a bridge apron, a flood gate and a pumping station. The lowest increase in predicted mean inanga abundance occurs along the midstream control sub-site and the downstream culvert sub-site, although zero predicted inanga were reported along upstream flood gate, pumping station and PS/ FFFG sub-sites, where no inanga were collected in situ and the treatments acted as upstream migratory barriers, as previously discovered in this study. These findings show that in this study, inanga inhabited warmer waterways, assuming that passage was unrestricted by a treatment which acts as a barrier.

A 1 SD increase in mean dissolved oxygen saturation leads to a significant IRR decrease in inanga abundance of 0.31 (95% CI: 0.14 − 0.69) across each position along all treatment types, compared to the intercept (Table 4.15). As with the 'Inanga Abundance Vs Barrier Status' dataset above, the predicted mean abundance of inanga against DO is in direct contrast to the findings of the mean temperature, where the most considerable decreases in inanga abundance occurs between 30-50% DO along upstream control and culvert sub-sites, the midstream PS/ FFFG sub-site and the downstream bridge apron, flood gate and pumping station sub-sites. This shows that the abundance of inanga decreases as DO saturation increases across all treatment types and that DO and temperature are directly correlated with one another (Fig.4.11).

Specific conductivity does not have a significant effect on the inanga abundances across all treatments, despite an increased inanga abundance IRR value of 1.56 (95% CI: 0.64 − 3.79) for every 1 SD increase in mean SPC, compared to the intercept (Table 4.15). Observing the EMM plot (Fig.4.12), the predicted mean inanga abundances are relatively low for all positions along all treatments below 10,000 µS/cm, though rises as the SPC increases, although the difference in abundance was greatest for upstream control and downstream bridge apron sites. However, non-significant differences indicate that specific conductivity does not have a considerable effect on the abundance of inanga across waterways with varying treatment.

The marginal R^2 value indicates that only 18.76% of the model's total variance is accounted for when the fixed environmental effects are used, though when the nested treatment and position random effects are included, the conditional R^2 value rises to 98.93%, showing the importance of incorporating both of the fixed and the random effects to account for all of the variance associated with the inanga abundance (Table 4.15).
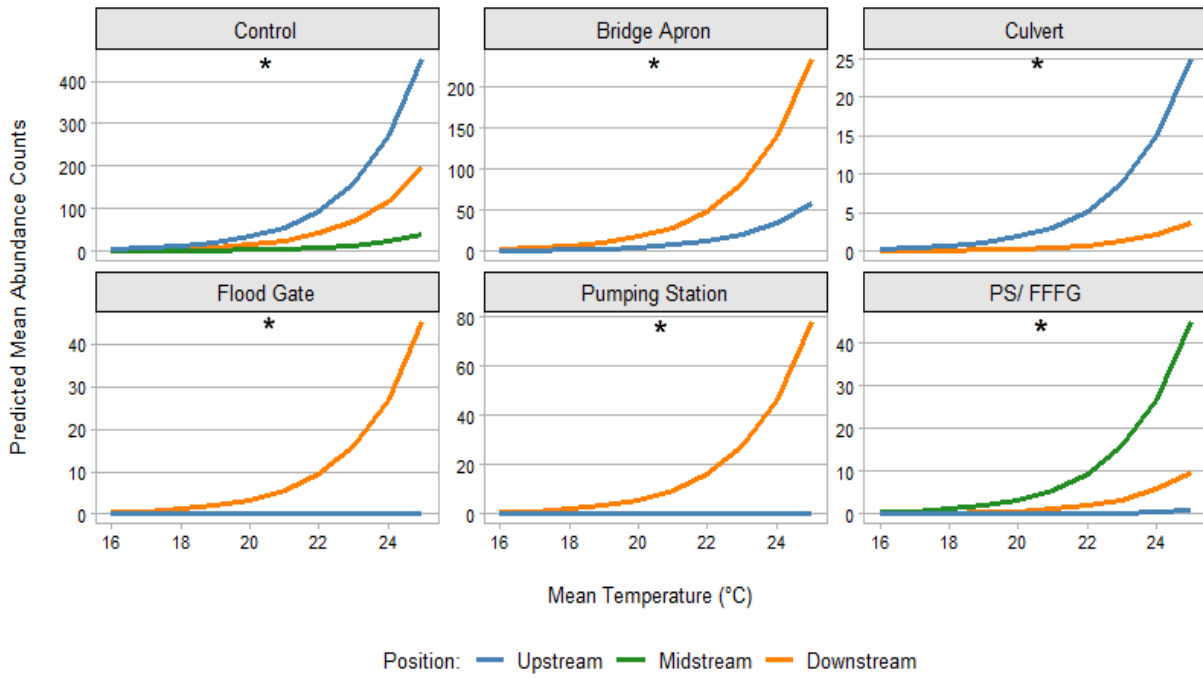
Figure 4.10. Estimated Marginal Mean (EMM) plots of predicted inanga mean abundance counts against mean temperature (°C) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
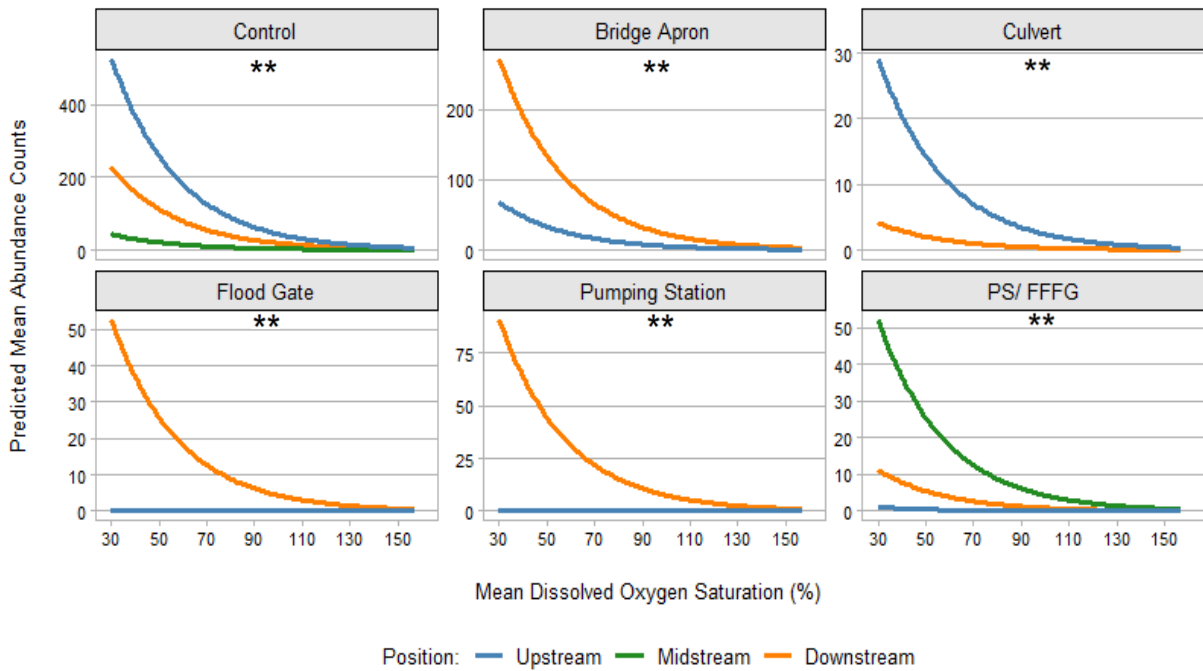
Figure 4.11. Estimated Marginal Mean (EMM) plots of predicted mean inanga abundance counts against mean dissolved oxygen saturation (%) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
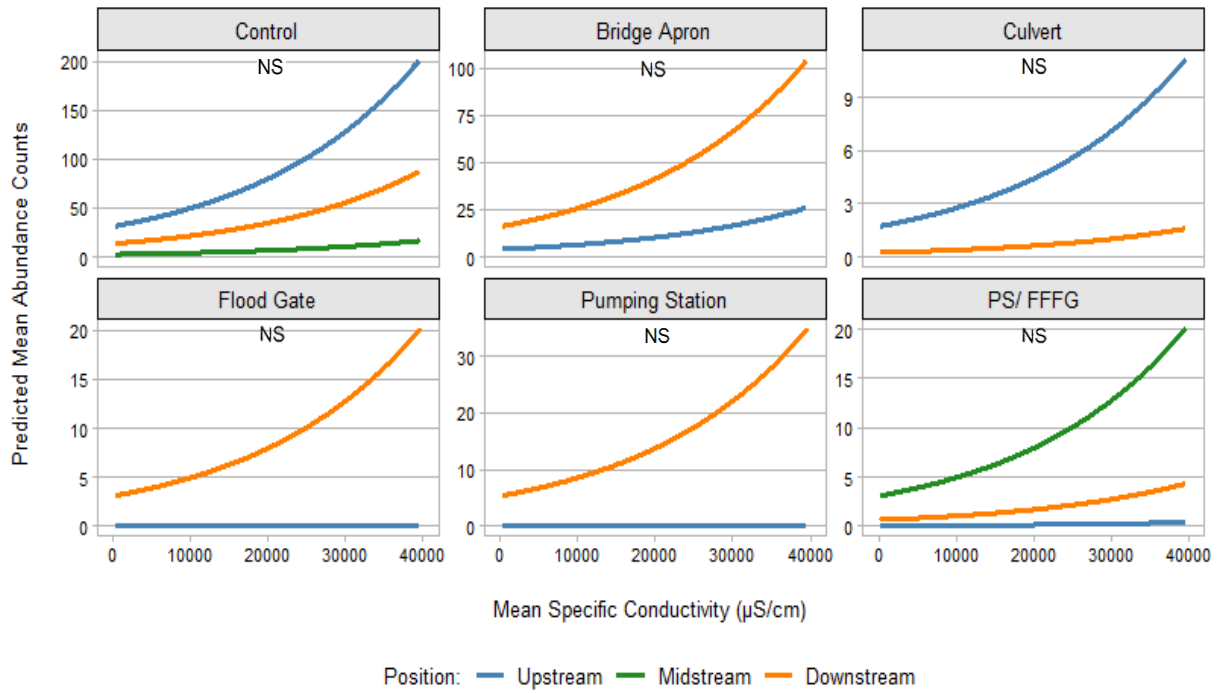
Figure 4.12. Estimated Marginal Mean (EMM) plots of predicted mean inanga abundance counts against mean specific conductivity (μS/cm) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).

Table 4.15. Estimated coefficients, their significance, (standard errors), z values and p-values for the scaled fixed effects and the variance and standard deviation (SD) for the nested random effects of a negative binomial GLMM using the 'Inanga Abundance Vs Treatment' dataset. SE = Standard Error, Temp = Temperature (°C), DO = Dissolved Oxygen Saturation (%), SPC = Specific Conductivity (μS/cm). Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Fixed Effects				
Scaled Covariate	Coefficient	*SE*	z value	p-value
(Intercept)	0.40	(0.86)	0.47	0.64
Mean Temp	1.22*	(0.49)	2.47	1.36E-02
Mean DO	-1.17**	(0.41)	-2.86	4.23E-03
Mean SPC	0.44	(0.45)	0.97	0.33

Random Effects			
Group	Name	Variance	SD
Treatment: Position	(Intercept)	8.59	2.93
Treatment	(Intercept)	1.23E-06	1.11E-03
Residual		0.75	0.57

Marginal R^2/ Conditional R^2		0.19/ 0.99	

4.4.5. All Species Total Abundance Vs Barrier Status

H_0: *Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the total abundance of all species along waterways with varying 'barrier status'.*

H_1: *Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the total abundance of all species along waterways with varying 'barrier status'.*

With scaled environmental fixed effects and nested barrier status and position random effects, the total abundance of all species significantly differs as the mean temperature, the mean dissolved oxygen saturation and the mean specific conductivity increases, rejecting the null hypotheses in favour of the alternatives that each of the three environmental variables have an effect on the total abundance of all species.

The total abundance of all species significantly increases by an IRR value of 1.25 (95% CI: 1.00 – 1.57) for every 1 SD increase in the mean temperature, compared with an intercept IRR value of 2.24 (95% CI: 1.23 – 4.07) whilst holding all other variables constant (Table 4.16). Observing the EMM plot (Fig.4.13), the predicted mean all species abundance increases gradually as the mean temperature increases across all positions and barrier statuses, with the upstream and the downstream positions showing similar rises across all barrier statuses. The midstream control sub-site shows little change in all species abundance as the temperature increases, although the all species abundance along the midstream non-barrier sub-site increases at a similar rate to the upstream and the downstream control and barrier waterways (Fig.4.13). These findings show that a higher temperature is correlated with increased all species total abundance, although the difference in abundances between sub-site positions is less than the difference observed between positions in the previous inanga analyses.

A 1 SD increase in mean dissolved oxygen saturation leads to a significant IRR decrease in all species total abundance of 0.71 (95% CI: 0.56 – 0.91), compared to the intercept (Table 4.16). With the highest EMM predicted mean all species abundance counts accumulating at 30% DO (Fig.4.14), each position across all barrier statuses showed a decrease in abundance as the mean dissolved oxygen saturation increased. In contrast to the mean temperature, the all species abundances decreases at a gradual and similar rate for the upstream and the downstream positions along control, barrier and non-barrier waterways, whilst the all species abundance along the single sampled midstream control sub-site did not considerable decrease as the mean DO saturation increased.

Specific conductivity has the greatest effect on the total abundance of all species as a 1 SD increase in SPC leads to a significant IRR increase of 1.92 (95% CI: 1.58 – 2.32) for all positions along all barrier statuses, compared with the intercept (Table 4.16). EMM plots indicate that the predicted mean all species abundances increased exponentially as the mean SPC increased for the upstream and the downstream control and barrier sites and the midstream non-barrier sites, whilst more gradual increases in abundances were predicted along the midstream control and the upstream and the downstream non-barrier positions (Fig.4.15). These findings show that a higher abundance of individuals across all species inhabit waterways with higher specific conductivity and that the total abundance of all species is similar across differing positions as the mean SPC increases.

The model's conditional R^2 value indicates that 79.61% of the all species total abundance variance is accounted for once both of the fixed and the random effects are included, considerably larger than the marginal R^2 value of 34.61% which explains the variance accounted for by the fixed effects only (Table 4.16).
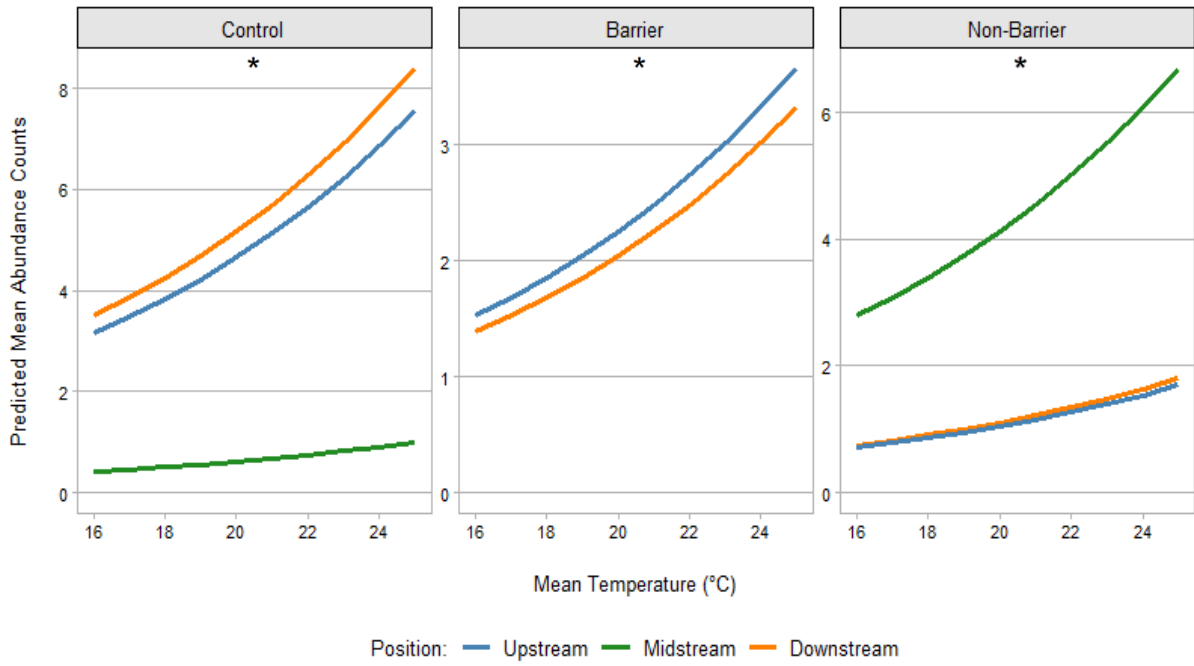
Figure 4.13. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean temperature (°C) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
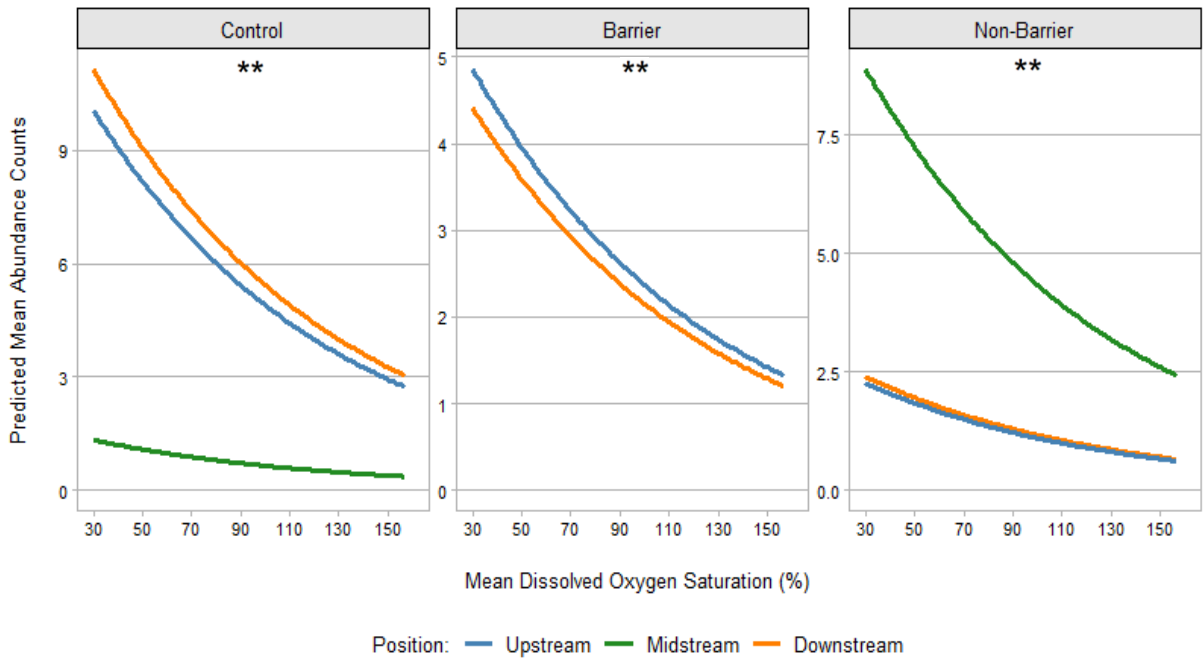
Figure 4.14. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean dissolved oxygen saturation (%) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
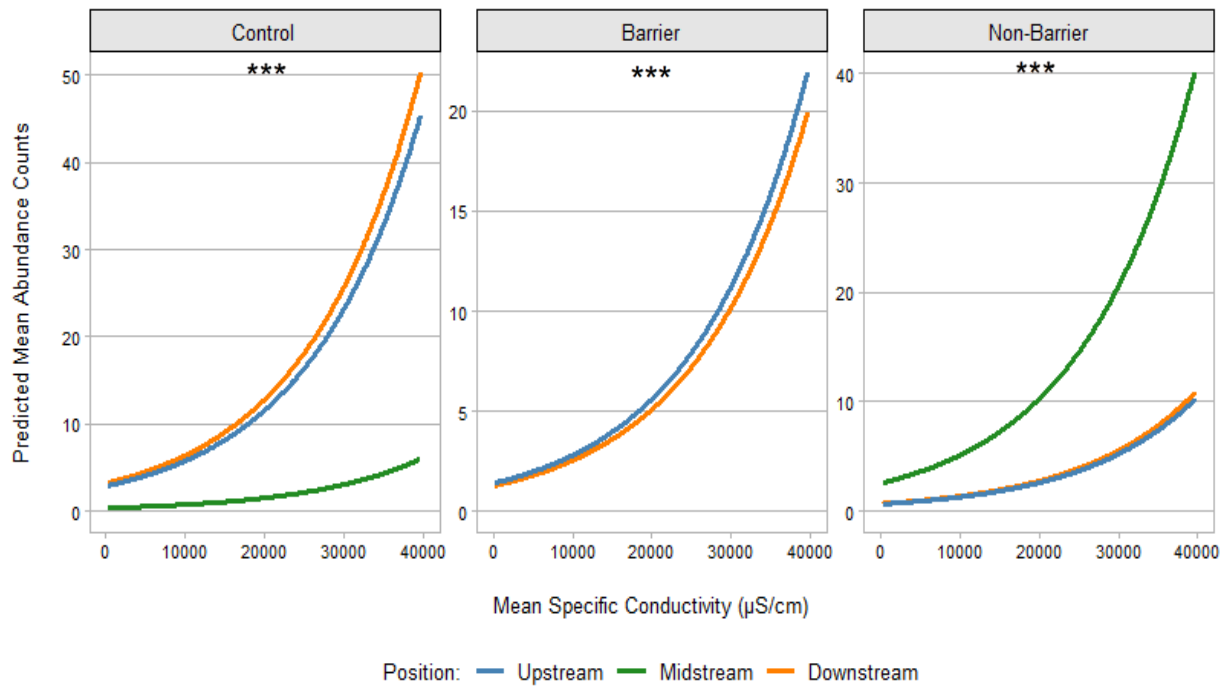
Figure 4.15. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean specific conductivity (µS/cm) for upstream, midstream and downstream positions, by barrier status. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).

Table 4.16. Estimated coefficients, their significance, (standard errors), z values and p-values for the scaled fixed effects and the variance and standard deviation (SD) for the nested random effects of a negative binomial GLMM using the 'All Species Total Abundance Vs Barrier Status' dataset. SE = Standard Error, Temp = Temperature (°C), DO = Dissolved Oxygen Saturation (%), SPC = Specific Conductivity (µS/cm). Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Fixed Effects				
Scaled Covariate	Coefficient	*SE*	z value	p-value
(Intercept)	0.81**	(0.30)	2.65	8.08E-03
Mean Temp	0.23*	(0.11)	1.98	4.83E-02
Mean DO	-0.34**	(0.12)	-2.78	5.46E-03
Mean SPC	0.65***	(0.10)	6.68	2.44E-11

Random Effects			
Group	Name	Variance	SD
Barrier Status: Position	(Intercept)	0.62	0.79
Barrier Status	(Intercept)	4.99E-08	2.23E-04
Residual		0.52	0.27

Marginal R^2/ Conditional R^2		0.35/ 0.80	

4.4.6. All Species Total Abundance Vs Treatment

H_0: *Temperature, dissolved oxygen saturation and specific conductivity will not have an effect on the total abundance of all species along waterways with varying 'treatment'.*

H_1: *Temperature, dissolved oxygen saturation and specific conductivity will have an effect on the total abundance of all species along waterways with varying 'treatment'.*

After scaling the environmental fixed effects and considering the variance of the nested treatment and position random effects, the total abundance of all species did not significantly differ as the mean temperature increased along waterways with varying treatment. In contrast, the total abundance of all species did significantly differ as the mean dissolved oxygen saturation and the mean specific conductivity increased along waterways with varying treatment.

A 1 SD increase in mean temperature leads to a total abundance of all species IRR value of 1.30 (95% CI: 0.98 – 1.74), compared to an intercept IRR value of 1.99 (95% CI: 1.31 – 3.02), although the difference is not significant (Table 4.17). Observing the EMM plot, the predicted mean all species abundance gradually increases for each position across all treatments as the mean temperature increases, although the rise in abundance along the midstream control sub-site is minimal (Fig.4.16). The all species abundances along the upstream and the downstream positions are similar across all treatments, indicating that the predicted abundances gradually increase as the temperature increases, although there is little difference between positions across all six treatments.

In contrast, a 1 SD increase in mean dissolved oxygen saturation leads to a total abundance of all species IRR value of 0.66 (95% CI: 0.50 – 0.87) compared with the intercept, indicating that the all species abundance significantly decreases as the DO saturation increases across all positions and along all treatments (Table 4.17). As with the temperature analyses, the all species total abundances along the upstream and the downstream positions are similar across all treatments, although the midstream control sub-site shows little change in predicted abundance (Fig.4.17). These findings show that the total abundance of all species is highest in waterways where the dissolved oxygen saturation is at its lowest in this study, irrespective of their treatment.

As with the 'All Species Total Abundance Vs Barrier Status' analysis above, the mean specific conductivity had the greatest effect on the total abundance of all species as a 1 SD increase led to an IRR increase of 1.96 (95% CI: 1.57 – 2.45) for all positions and across all treatments, compared to the intercept (Table 4.17). The predicted mean all species abundance on the EMM plot increased at similar rates for the upstream and the downstream positions of each treatment as the mean SPC increased, whilst the abundance of the midstream control sub-site increased only marginally, and the abundance of the midstream PS/ FFFG sub-site increased to a greater extent (Fig.4.18), indicating that the total abundance of all species increases as the mean specific conductivity increases, although the difference between upstream and downstream positions is minimal.

Examining the marginal and the conditional R^2 values, only 41.02% of the abundance is accounted for by the fixed effects, though once the nested treatment and position random effects are included, 70.58% of the variance is considered, indicating that the model accounts for a larger amount of the dataset's variance once both of the fixed and the random effects are included (Table 4.17).
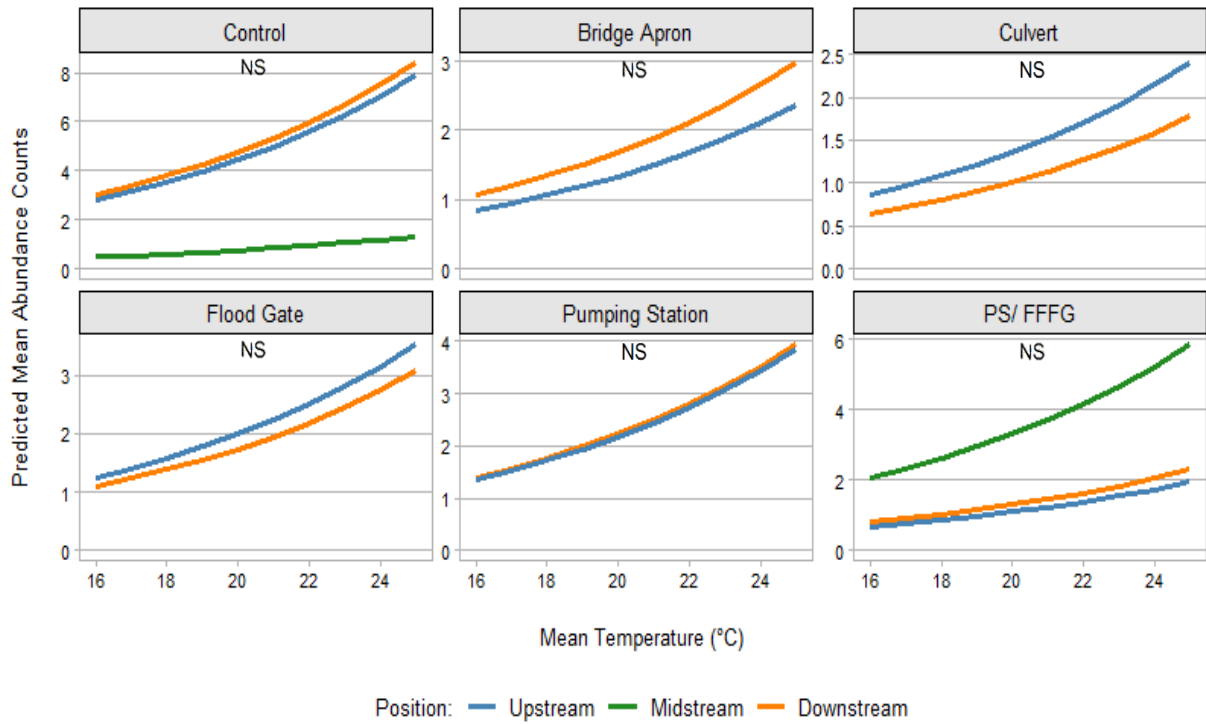
Figure 4.16. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean temperature (°C) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
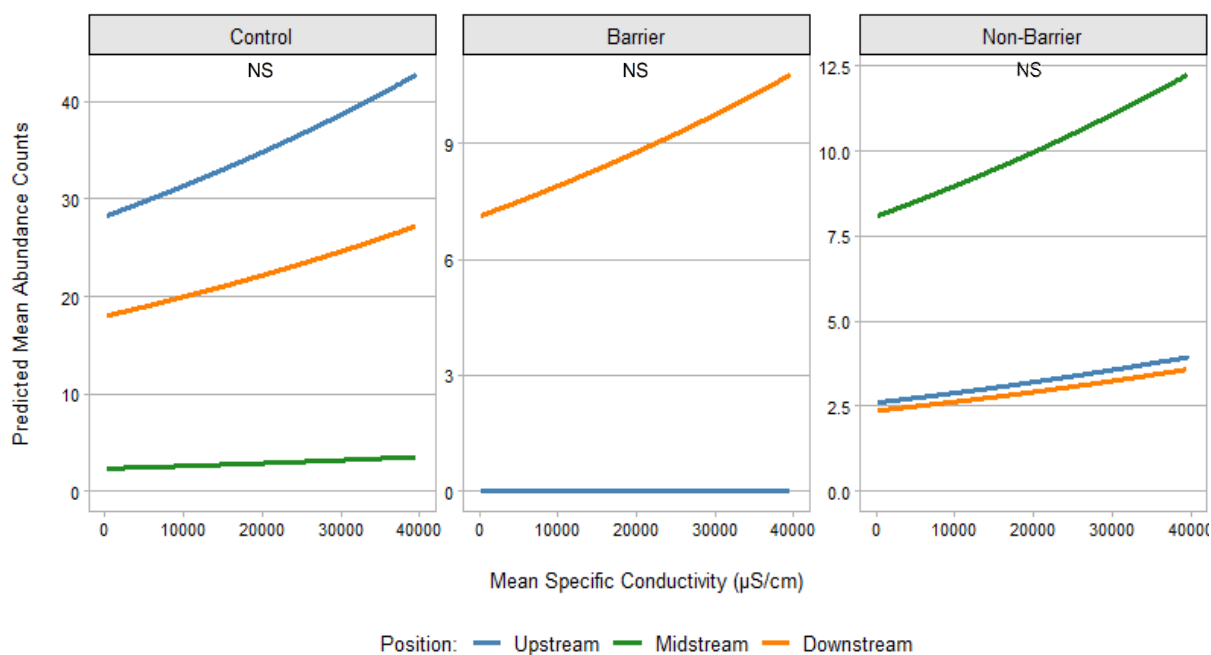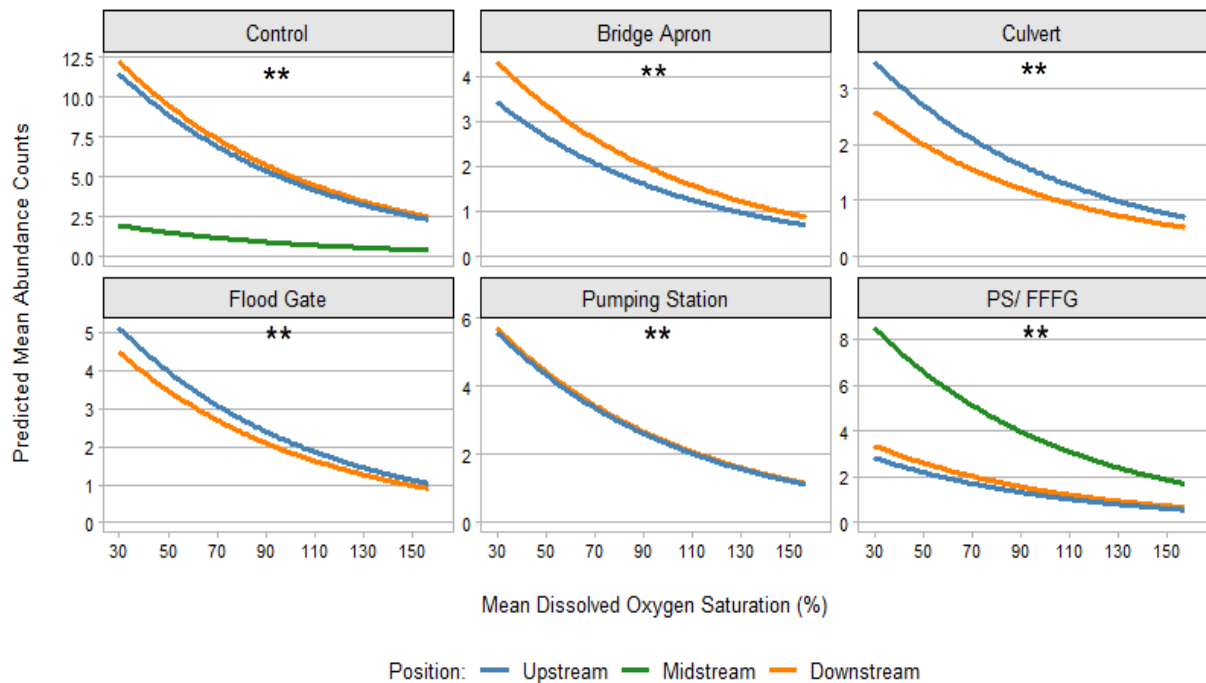

Figure 4.17. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean dissolved oxygen saturation (%) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).
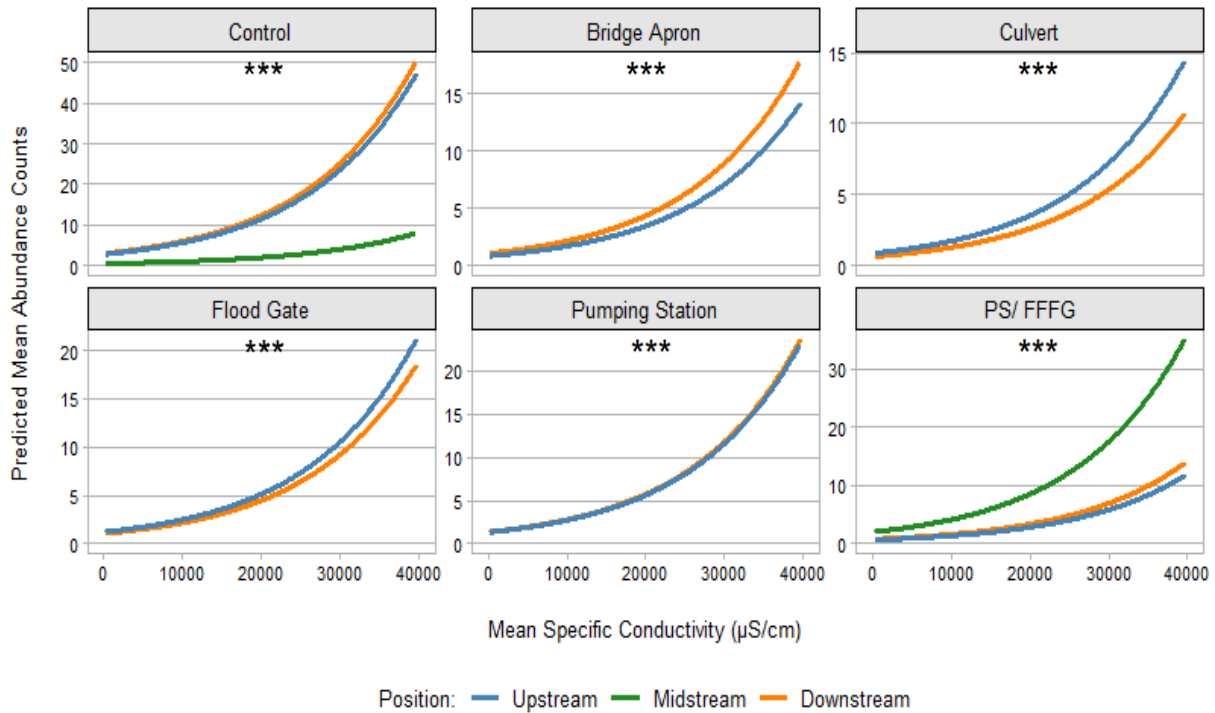
Figure 4.18. Estimated Marginal Mean (EMM) plots of predicted mean all species abundance counts against mean specific conductivity (µS/cm) for upstream, midstream and downstream positions, by treatment. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLMM).

Table 4.17. Estimated coefficients, their significance, (standard errors), z values and p-values for the scaled fixed effects and the variance and standard deviation (SD) for the nested random effects of a negative binomial GLMM using the 'All Species Total Abundance Vs Treatment' dataset. SE = Standard Error, Temp = Temperature (°C), DO = Dissolved Oxygen Saturation (%), SPC = Specific Conductivity (µS/cm). Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05.

Fixed Effects				
Scaled Covariate	Coefficient	*SE*	z value	p-value
(Intercept)	0.69**	(0.21)	3.24	1.21E-03
Mean Temp	0.27	(0.15)	1.83	6.80E-02
Mean DO	-0.42**	(0.14)	-2.95	3.16E-03
Mean SPC	0.67***	(0.11)	5.92	3.29E-09

Random Effects			
Group	Name	Variance	SD
Treatment: Position	(Intercept)	0.39	0.63
Treatment	(Intercept)	3.86E-08	1.96E-04
Residual		0.52	0.27

Marginal R^2/ Conditional R^2	0.41/ 0.71

4.4.7. Summary of Environmental Results

Table 4.18. Summary of the hypotheses, significance of results and the context at which the results can be applied in situ for the environmental analyses of the 'Inanga Abundance Vs Barrier Status', 'Inanga Abundance Vs Treatment', 'All Species Total Abundance Vs Barrier Status' and the 'All Species Total Abundance Vs Treatment' datasets. Temp = Temperature (°C), DO = Dissolved Oxygen Saturation (%), SPC = Specific Conductivity (μS/cm).

Inanga Abundance Vs Barrier Status

Hypotheses	Covariate	Significance	Context
H_0: Temp, DO and SPC will not have an effect on the abundance of inanga along waterways with varying 'barrier status'.	Temp (°C)	Sig. Different	Inanga abundance **higher** as temperature increases
	DO (%)	Sig. Different	Inanga abundance **lower** as dissolved oxygen saturation increases
H_1: Temp, DO and SPC will have an effect on the abundance of inanga along waterways with varying 'barrier status'.	SPC (μS/cm)	Not Sig. Different	Inanga abundance **NOT changed** as specific conductivity increases

Inanga Abundance Vs Treatment

Hypotheses	Covariate	Significance	Context
H_0: Temp, DO and SPC will not have an effect on the abundance of inanga along waterways with varying 'treatment'.	Temp (°C)	Sig. Different	Inanga abundance **higher** as temperature increases
	DO (%)	Sig. Different	Inanga abundance **lower** as dissolved oxygen saturation increases
H_1: Temp, DO and SPC will have an effect on the abundance of inanga along waterways with varying 'treatment'.	SPC (μS/cm)	Not Sig. Different	Inanga abundance **NOT changed** as specific conductivity increases

All Species Total Abundance Vs Barrier Status

Hypotheses	Covariate	Significance	Context
H_0: Temp, DO and SPC will not have an effect on the total abundance of all species along	Temp (°C)	Sig. Different	Total abundance of all species **higher** as temperature increases

| *waterways with varying 'barrier status'.* | DO (%) | Sig. Different | Total abundance of all species **lower** as dissolved oxygen saturation increases |
| *H₁: Temp, DO and SPC will have an effect on the total abundance of all species along waterways with varying 'barrier status'.* | SPC (µS/cm) | Sig. Different | Total abundance of all species **higher** as specific conductivity increases |

All Species Total Abundance Vs Treatment

Hypotheses	Covariate	Significance	Context
H₀: Temp, DO and SPC will not have an effect on the total abundance of all species along waterways with varying 'treatment'.	Temp (°C)	Not Sig. Different	Total abundance of all species **NOT changed** as temperature increases
H₁: Temp, DO and SPC will have an effect on the total abundance of all species along waterways with varying 'treatment'.	DO (%)	Sig. Different	Total abundance of all species **lower** as dissolved oxygen saturation increases
	SPC (µS/cm)	Sig. Different	Total abundance of all species **higher** as specific conductivity increases

Chapter 5

Discussion

The detailed methodology, the results and the following discussion sections have ensured that the aims and the objectives of this study, outlined in section 1.6., have been achieved. Fish populations, with a particular focus on inanga, and site-specific environmental variables along 16 lowland waterways across the Hawke's Bay region of New Zealand were meticulously sampled and described, and the yielding data was statistically analysed and evaluated to assess the level of impact of each assessed anthropogenic drainage structure.

Using the findings described in the results section of this study, the discussion will address the third study objective pre-mentioned in section 1.6., by looking into the context at which the results can be applied in a real-world setting and will subsequently make management recommendations. The passability of a bridge apron, a culvert, flood gates, a FFFG and pumping stations on the upstream migrations of inanga and other present species will be discussed by considering the differences in abundances and biodiversity within upstream, midstream, and downstream sub-sites. Findings will be compared to relevant studies within the literature and the effectiveness of the control sites used in this study will be assessed. The effects of temperature, dissolved oxygen saturation and specific conductivity on fish abundances will be discussed, along with recommendations for potential barrier mitigations and site remediations. The discussion concludes by addressing the limitations of the study and recommends additional further research which could build on the findings of this study.

5.1. The Physical Effects of Barriers on Fish Abundance and Biodiversity

The National Fish Passage Symposium of 2014 defined a fish migration barrier as a structure, whether natural or anthropogenic, that limits or impedes the upstream or downstream movement of a species, resulting in the decline of native fish abundances, the reduction of an ecosystem's biodiversity and the potential localised extinctions of entire populations (Franklin, 2014). The identification of large-scale barriers, like hydroelectric dams are often uncomplicated due to their size, though classifying small-scaled drainage structures, like those assessed in this study may not be as straight-forward (Franklin, 2014). A structure's fish passability and therefore its barrier status differs according to a number of factors, primarily the ecology and the life history of each specific species, though secondary impediments like environmental or habitat alterations caused by the structure (Franklin, 2014) can lead to further migratory restrictions and contribute towards its barrier status.

It is well documented within the literature that inanga exhibit an amphidromous life cycle, where juveniles, or 'whitebait' drift upstream from salt to freshwaters to feed and sexually mature into adult inanga (Franklin and Gee, 2019). Adults spawn within the riparian vegetation along streambanks close to the freshwater-saltwater boundary zone, before newly hatched larvae drift back downstream into marine habitats to grow for a 3-6 month period (McRae et al., 2016).

Amphidromy as a life cycle depends largely on the unrestricted upstream passage of juvenile and adult fish alike, rendering them highly vulnerable to barrier impacts from anthropogenic drainage structures which often lie along, or close to suitable inanga spawning habitat (McRae *et al.*, 2016). With weak swimming and climbing abilities (Stevens *et al.*, 2016), inanga are often unable to navigate beyond small-scale structures, like bridge aprons and culverts, though larger-bodied adult individuals may be able to pass if conditions do not exceed their swimming abilities. If structures like flood gates and pumping stations span the width of a water channel, upstream migrations may become entirely restricted for all inanga, regardless of their size or swimming ability.

However, inanga with amphidromous life cycles are not exclusively impacted by small-scale anthropogenic structures. Larger-bodied fish observed in this study, like grey and yellow-eye mullet, and shortfin and longfin eels, have catadromous life cycles, where juveniles migrate upstream into freshwaters to feed and sexually mature, before adults migrate back downstream into saltwaters to spawn (Lassalle *et al.*, 2008). Although exhibiting stronger swimming and climbing abilities than inanga (Stevens *et al.*, 2016), upstream migrating juvenile catadromous fish may still succumb to the impacts of anthropogenic barriers, preventing access to crucial, available upstream habitat required to feed, rest and grow. Some species however may form resident populations in upstream habitats above a barrier, like mosquitofish and goldfish as observed in this study, though are often invasive, do not require a migratory life cycle to survive and are more tolerant to less favourable environmental conditions, rendering them unimpacted by anthropogenic drainage structures.

Although this study analyses the passability of specific anthropogenic structures at the freshwater-saltwater interface along the lowland waterways of Hawke's Bay, New Zealand, the applicability of the findings and subsequent suggestions for remediation are far-reaching and can be applied to comparable drainage structures and similar fish species across the world.

5.1.1. Control Sites

Studied sites: Upper Taipo stream, lower Taipo stream, Grange Creek, Te Ngarue stream, Pakuratahi stream.

In order to reliably compare the effect of anthropogenic drainage structures on the abundance and biodiversity of fish along the lowland waterways of Hawke's Bay, five control sites, where no structure was present, were used and compared against. The above control sites showed what would be expected if full upstream passage for diadromous fish was unrestricted by a structure. The five sites were deemed effective and appropriate for the experimental sites in this study as inanga were observed within all upstream, midstream and downstream control sub-sites, other than within the downstream Te Ngarue stream sub-site (Table 4.1). Although inanga abundances differed considerably between control waterways, sub-populations were observed within all upstream sub-sites, showing that given the opportunity, inanga will migrate into upstream freshwater environments in an attempt to find suitable feeding, resting and spawning habitat. Therefore, if inanga are present within downstream sub-sites where a structure is present, but not within the upstream sub-sites, then that structure does restrict upstream passage and acts as a migratory barrier.

As many of the structures assessed in this study lie along the freshwater- saltwater boundary zone, which exhibits unique environmental and habitat qualities and is typically associated with a higher species richness, it was important to also sample downstream control sub-sites which lie along, or close to similar boundary zones. Therefore, the species richness and the biodiversity within the

waterways where a structure was present could be reliably compared to the richness and the biodiversity within the control waterways where no structure was present. Lower upstream measures above a barrier would suggest that upstream passage is restricted by the structure, rather than due to differing sampling locations along a stream.

5.1.2. Bridge Apron

Studied site: Mangarau stream.

The abundance of inanga (Fig.4.2; Table 4.3) and the total abundance of all species (Fig.4.4; Table 4.3) were lower in the upstream sub-site above the bridge apron, compared with the abundances downstream below the bridge apron, or compared to the abundances found within the control sites. However, one inanga individual was found inhabiting the upstream sub-site, indicating that upstream fish passage is not fully restricted by the bridge apron. Similar species richness, species evenness and non-significantly different biodiversity measures between the upstream and the downstream sub-sites (Fig.4.6; Table 4.3) further indicate that the ~30cm vertical drop of the bridge apron does not act as a full barrier, but instead is a partial upstream migratory barrier to small-bodied diadromous fish. These findings are partly consistent with a study by Baker (2003) who found that the migrations of weaker swimming juvenile inanga and common bullies were restricted by a weir drop height of 10cm, though larger bodied adult inanga with greater burst-swimming capabilities were able to negotiate higher water velocities (Behlke *et al.,* 1991) and continue upstream beyond a drop height of up to 20cm. Although Baker (2003) assessed the passability of a weir, findings are comparable with this study's bridge apron as the drop heights and the level of turbulence and water velocities are similar across both structures. The single upstream adult inanga in this study may have gained access beyond the ~30cm bridge apron drop during a high rainfall event, where the stream's water levels rose and the height of the drop was reduced to a similar level as that proven to be accessible for inanga in the Baker (2003) study, an observation also recorded by Winter and Van Densen (2001).

A similar study by Fake (2018) found that a perched apron with a drop measuring ~70cm along the Waingongoro stream, New Zealand, acted as a full migratory barrier to swimming species like the torrentfish *(Cheimarrichthys fosteri)*, a small-bodied amphidromous fish with similar burst swimming abilities as inanga, as overall abundances and species richness were significantly lower within the upstream sub-site, whereas species with greater climbing capabilities like eel sp. and redfin bullies *(Gobiomorphus huttoni)* were able to migrate above the structure. The difference in barrier statuses between this study and that by Fake (2018) is associated with the greater drop height observed beneath the Waingongoro stream's perched apron, signifying that slight alterations in a structure's physical dimensions can have detrimental effects on migrating species with weaker swimming or climbing abilities.

This study's results support the assertions by European studies that the passability of a structure with a vertical drop, like a weir, varies according to a migrating individual's swimming capabilities or body size. A long-term study by Winter and Van Densen (2001) reported that several 10cm length, weak swimming gibel carp *(Carassius auratus gibelio)* were unable to ascend any of the six studied weirs of various heights along the River Vechdt, the Netherlands, whilst fast swimming dace *(Leuciscus leuciscus)* of the same size were able to migrate above the structure. In addition to the swimming abilities of fish, Lucas and Frear (1997) found that only the largest six of the 23 tracked common barbel *(Barbus barbus)* successfully passed a 40cm weir drop along the River Nidd, UK, with

those that did not pass the structure experiencing disrupted or delayed courtship and subsequent spawning success. Barbel typically range from 25-100cm in length (Lucas and Frear, 1997), vastly greater than the average adult inanga length of 10cm (Baker, 2003) signifying that if only 26% of large-bodied barbel are able to migrate above a drop height of 40cm, it is no surprise that only 17% of inanga in this study were able to negotiate a drop of ~30cm.

An increasing amount of literature in recent years has focused on the positive effects that an installed fish ramp can have on the passage of migrating fish above anthropogenic structures, with the ramp efficacy now being recognised to be influenced by its positioning (Baker, 2003; Baker and Boubée, 2006), the angle at which the ramp is installed (Baker and Boubée, 2006; Doehring *et al.,* 2012) and the type of additional retrofitting built onto the ramp (Baker and Boubée, 2006; Fake, 2018). A study by Baker and Boubée (2006) found that over 75% of juvenile inanga, or whitebait, and over 90% of adult inanga were able to negotiate all of the assessed installed fish ramps, assuming that the angle of the ramp's slope was at 15°, as slopes greater than this value caused downstream water velocities which exceeded the inanga's weak swimming abilities. Fake (2018) installed a plastic ramp lined with small cusps and an attached spat rope retrofit onto a vertical drop beneath a large culvert along the Awanui stream, a similar drop as the one assessed in this study and one which was also observed to act as a migratory barrier. Results showed that the abundance of inanga upstream of the structure pre-ramp installation was significantly different to the abundance post-installation, indicating the success of the ramp in providing upstream passage for the small-bodied amphidromous fish.

Figure 5.1. Plastic fish ramp, lined with small cusps, similar to the one used in Fake (2018), installed onto the vertical drop of the bridge apron along the Mangarau stream.

A similarly designed fish ramp was then acquired for installation on the Mangarau stream bridge apron's vertical drop (Fig.5.1) in an attempt to also improve fish passage into the upstream habitat. A ramp lined with small cusps was used which provides lower velocity resting spaces for weaker swimmers and wetted margins for climbing species to successfully continue upstream (Fake, 2018). Currently, no further data has been collected to assess the passability of this study's now remediated bridge apron drop, though further research is required to assess whether the installation of the ramp has led to improved upstream passage for migrating fish.

Studied site: Ngaruroro Backwash.

The abundance of inanga is significantly higher within the upstream sub-site above the pipe culvert than the abundance within the downstream sub-site below the pipe culvert or compared to those within the control sites (Fig.4.2; Table 4.3), indicating that the drainage structure does not act as an upstream migratory barrier to this study's focal amphidromous fish. While a greater total abundance of all species was encountered upstream compared with the downstream, this difference was not significantly higher (Fig.4.4; Table 4.3), though significantly different biodiversity measures above and below the culvert (Fig.4.6; Table 4.3) signifies that a partial or temporary barrier exists for some fish inhabiting the stream.

The concept of a culvert acting as a partial barrier, as in this study, has received some attention within the literature. Doehring *et al.* (2011a) investigated the difference in passability between perched culverts, or those where the structure's outlet lies above the water surface and non-perched culverts, where the outlet remains partially-fully submerged within the waterway. The study found that 0% of small-bodied juvenile inanga migrated beyond a perched culvert when the outlet's fall height lay between 0.5-9.5cm, potentially offering an explanation as to how the ~30cm fall height beneath the culvert in this study during a low tide may act as a temporary barrier to fish with weak swimming, climbing and jumping albitites. Doehring *et al.* (2011a) further described how 65% of fish were able to pass non-perched, partially-fully submerged culverts, explaining how upstream passage was temporarily possible for some migrating species in this study as the rising downstream water levels fully submerged the culvert outlet during a high tide.

This study's assertions that a culvert temporarily allows, or restricts, upstream passage and subsequently affects the stream's overall fish abundance and species biodiversity, are similar to existing literature in the USA studying the effect of culverts on the weak swimming, small-bodied creek chub *(Semotilus atromaculatus)* (e.g., Nislow *et al.,* 2011; Briggs and Galarowicz, 2013). They found that fully passable structures exhibited similar fish abundances and species richness between upstream and downstream sub-sites, whereas passage was considerably affected when a culvert's physical dimensions exceeded the target species' swimming albitites, i.e., if a perched culvert's outlet drop was too high and subsequent water flow through the structure was too strong (Anderson *et al.,* 2012). In contrast, each of the above studies found that even for temporary or partial barriers, species richness was significantly lower upstream compared with downstream, whereas there was no difference in either species richness or evenness between sub-sites in this study. These findings indicate that fish of all species, whether migratory or resident, are able to navigate beyond the culvert into the upstream sub-site during a high tide, even if their abundances differ between sub-sites and that the culvert along the Ngaruroro Backwash acts as a lesser barrier than those assessed in previous studies.

Whether the culvert is permanently, or temporarily perched during a low tide, as is the case in this study, the velocity of the downstream water flow can often act as an additional migratory barrier for juvenile or weak swimming adult fish, including this study's focal inanga (House *et al.,* 2005). Franklin and Bartels (2012) found that a community of small-bodied migrating fish, including inanga and common bullies along the Bankwood stream in Waikato, New Zealand, were unable to migrate beyond a 74m long culvert, despite the presence of an installed baffled fish ramp, as the hydraulic conditions within the structure exceeded the individual's 'burst' swimming capabilities (Behlke *et al.,* 1991), which for inanga reaches speeds of 0.34m/s^{-1} for >30s or 0.19m/s^{-1} for 20 minutes of sustained swimming (Mitchell, 1989). However, as also noted by MacDonald and Davies (2007), once

spoiler baffles were installed along the base of the culvert, the downstream water velocity lowered and refuge space for weaker swimming fish between the baffles became available, allowing inanga to successfully migrate upstream (Franklin and Bartels, 2012). Although it is assumed that the velocity and subsequent turbulence observed during a low tide in this study acts as a hydrological barrier to migrating fish, additional research may be required in order to assess the true passability of the temporarily perched culvert so that the most effective remediations can be implemented to allow upstream access for weak swimming species at all tidal periods.

Given the vast number of culverts which fragment low order streams and rivers globally (Liermann *et al.*, 2012) and the often high costs associated with repairing, removing or replacing the structure, numerous practical, low cost fish passage techniques have been developed and assessed in the literature. As previously mentioned as a bridge apron remediation, fish ramps are a suitable and well analysed culvert retrofit due to their low unit cost compared with traditional fish passes, their naturally buoyancy so that its position can self-adjust during fluctuating water levels (Fake, 2018) and their potential for additional textured panels or baffles to be installed which are able to rapidly blend into the surrounding landscape (Baker and Boubée, 2006; Fake, 2018). However, regular ramp maintenance may be required along waterways with high macrophyte or woody debris drift, so that channel blockage does not occur, and passage is still possible (Fake, 2018), an issue which does not exist in this study. Studies by Doehring *et al.* (2011a) and Franklin and Bartels (2012) reported that the upstream passage of juvenile and adult inanga increased by 44% and 27%, respectively, when perched culverts were retrofitted by fish ramps, though different ramp and culvert lengths, gradients and slopes may have different effects on passage for different species and that restorations should be designed appropriately for target species.

Where space for fish ramps is limited, or installation of baffles is impractical, spat ropes offer an effective, low-cost method to facilitate upstream passage for small-bodied fish. David *et al*. (2014) found that successful inanga and New Zealand freshwater shrimp passage increased from 23% without ropes to 83% with ropes along a 3m length pipe culvert, whilst passage rose from 0% to 57% along a 6m long pipe culvert. Initial remediation of the culvert in this study should include the installation of a fish ramp with attached spat rope and a continued upstream fish abundance assessment with the aim of aiding additional upstream migration during a low tide.

5.1.4. Flood Gate

Studied sites: Pakowhai, Memorial Park Drain, Muddy Creek (FFFG divides the midstream and the downstream sub-sites), Tutaekuri Waimate stream (flood gate present, though unsuitable environmental conditions meant that data was not collected along this stream).

The abundance of inanga within upstream sub-sites above a flood gate were lower than the downstream abundances below a flood gate, or those within the control sites in this study (Fig.4.2; Table 4.3). The biodiversity, species richness and species evenness between sub-sites were significantly different, signifying that the flood gates assessed in this study do act as complete barriers to upstream migrating fish (Fig.4.6; Table 4.3). The total abundance of all species was significantly higher upstream, compared with the total abundance downstream, though the more environmentally tolerant New Zealand freshwater shrimp and the exotic mosquitofish account for the vast majority of these high upstream abundances, whilst other migratory species, like grey or yellow-eye mullet and common bully were encountered downstream only (Fig.4.6; Table 4.3; Table 4.10).

It has been widely accepted in the literature that flood gates act as partial barriers to migrating fish, though the degree at which a structure impedes passage is dependent on the characteristics of the gate, the upstream habitat and each species' swimming capabilities (Kroon and Ansell, 2006; Bocker, 2015; Franklin and Hodges, 2015). However, the findings of this study indicate that a barrier exists no matter how strong a fish's swimming abilities because individuals of all species, including the stronger swimming mullet sp. were all absent upstream of the assessed flood gates. In a similar experimental study, Doehring *et al.* (2011b) observed less than half as many fish in habitats above a flood gate than in sites where a structure was not present, though a population of juvenile inanga, mullet sp., New Zealand freshwater shrimp and eel sp. remained present within the upstream habitat. Doehring *et al.* (2011b) suggested that the velocity of the downstream flow through the structure during ebb tides did not exceed the swimming capabilities of any of the migratory fish encountered upstream, though excessive flows through this study's flood gates may provide an explanation as to why no migrating species were present upstream. It is recommended that further assessments analysing the water flow velocity through flood gates similar in size as those in this study should be completed to determine whether the structure acts solely as a physical barrier, a hydrological barrier (Franklin and Hodges, 2015), or a combination of the two.

Significantly lower upstream biodiversity and species richness measures reported in this study were also observed in a study by Pollard and Hannan (1994) and more recently by Bocker (2015), where a mean upstream richness of 6 species was dominated by a high abundance of the exotic mosquitofish and goldfish, whereas estuarine-marine species like grey and yellow-eye mullet were 70-80% more abundant within a downstream fish community comprising of 9 separate species, signifying that the flood gates in their studies, as well as those from this study do indeed restrict the upstream passage of ecologically important species. Flood gates also have the potential to act as significant environmental barriers, as discussed later in this chapter, which can often lead to the degraded, but optimal conditions for exotic and pest species to flourish. As also discovered in this study, Kroon and Ansell (2006) found that 23% of the total catch along streams where a flood gate was present comprised of the live-bearing mosquitofish, a species which can sustain populations in conditions that are often lethal for more sensitive fish to inhabit (Boys *et al.,* 2012; Franklin and Hodges, 2015), though they comprised only 0.69% of the total catch along higher quality, control sites. The dominance of invasive fish and large-bodied, environmentally tolerant species like shortfin or longfin eel encountered above the flood gate in this study often provides additional challenges for small-bodied migrating fish, like interspecific competition or increased predation pressure (Doehring *et al.,* 2011b; Bocker, 2015), even if upstream passage were possible.

Whilst discussing the techniques of facilitating fish passage across anthropogenic drainage structures, the literature historically prioritised culverts and weirs for remediation, with substantially less consideration given to flood gates, despite their common use across global lowland waterways (Bocker, 2015). Early studies, like that by Pollard and Hannan (1994), suggested that flood gates should be left permanently open, except during peak tidal times, where they should close once again and retain high water levels downstream. More recently however, and since it was realised that amphidromous fish migrate upstream primarily during these peak tidal times, the suggestions by Pollard and Hannan (1994) have been disregarded and the installation of 'fish-friendly' retrofit mechanisms onto existing flood gates has received considerable attention. Remediated 'fish-friendly flood gates', or FFFGs, increase the width and the duration of time that the gate is open for during a high tide (Bocker, 2015), providing additional time and more suitable conditions with less water velocity, turbulence and a more gradual transition between fresh and saltwaters for fish to migrate upstream (ref. Giannico and Souder, 2004 for specific FFFG model retrofits).

A single, recently remediated FFFG dividing the midstream and the downstream sub-sites along Muddy Creek were assessed in this study to determine whether passage for migrating fish is possible beyond a retrofitted flood gate. The abundance of inanga (Fig.4.2; Table 4.1) and the total abundance of all species (Fig.4.4; Table 4.1) were significantly higher within the midstream sub-site above the FFFG than the abundances within the downstream sub-site below the FFFG, or than the abundances within the control sites. However, the biodiversity, the species richness and the species evenness (Fig.4.6; Table 4.10) were similar between sub-sites and therefore yielded non significantly different results, signifying that the FFFG allows complete upstream passage and does not act as a barrier to migrating fish. These findings are consistent with those from studies by Boys *et al.* (2012), Bocker (2015) and Franklin and Hodges (2015) who found 20 times more juvenile and adult inanga and 4 times more common bully within upstream sub-sites when a remediated FFFG was present, compared to waterways without the 'fish-friendly' influence. Bocker (2015) discussed how a 45-minute increase in gate opening duration led to a significantly higher number of weak swimming species being able to navigate beyond the structure, whilst species richness measures between sub-sites became equal one year after FFFG installation, a finding comparable to this study as 6 separate species were encountered both above and below the remediated structure. In contrast however, Bocker (2015) mentioned that a significant decline in mosquitofish had been observed between pre and post installation assessments, though a considerable population of the same species was encountered above the FFFG in this study, potentially indicating that degraded or unfavourable environmental conditions remain enabling invasive species to continue to thrive, though the environmental effects of a FFFG remediation will be further discussed later in this chapter.

5.1.5. Pumping Station

Studied sites: County Drain, Old Tutaekuri Riverbed, Purimu stream, Ahuriri Station Drain, Plantation Drain, Muddy Creek (pumping station divides the upstream and the midstream sub-sites).

The abundance of inanga is lower in the upstream sub-sites above a pumping station than the inanga abundances downstream below a pumping station, or than the abundances within the control sites of this study (Fig.4.2; Table 4.3). The biodiversity, the species richness and the species evenness are significantly different between sub-sites, indicating that the pumping stations present along the above waterways act as full and complete barriers to upstream migrating fish (Fig.4.6; Table 4.10). The total abundances of all species were significantly higher above the structure, compared with those below the structure (Fig.4.4; Table 4.3), although the invasive mosquitofish and goldfish and the environmentally tolerant New Zealand freshwater shrimp and eel sp. account for the vast majority of species encountered above the pumping stations, whilst migratory fish were absent from all upstream sub-sites (Table 4.10).

The differences in fish community assemblages between sub-sites and the lack of any migratory fish (excluding eel sp.) above each pumping station in this study are consistent with the findings from other New Zealand studies by Joy and Death (2001) and Jellyman and Harding (2012) and international studies by Gehrke *et al.* (2002) in Australia and Katano *et al*. (2006) in Japan. They found that the abundance of diadromous fish, the biodiversity and the species richness above 1.5-3.9m high pumping stations were lower than the abundances, the biodiversity and the species richness below the structures or those along undammed, control waterways. Encountering species also assessed in this study, Joy and Death (2001) and Jellyman and Harding (2012) found that inanga, grey and yellow-eye mullet were all absent above the barriers and were found exclusively in downstream sub-sites, whereas the percentage of exotic species like mosquitofish and goldfish

within upstream communities were significantly higher than those downstream. The strong climbing and environmentally tolerant shortfin eel and the longfin eel were commonly encountered within the upstream sub-sites of this study and within those assessed by Jellyman and Harding (2012) due to their abilities to physically climb beyond a pumping station wall by scaling its wetted margins (Fake, 2018). However, due to their catadromous life cycle and requirement to migrate back to the ocean to breed, it has been widely reported that dams, or pumping stations act as significant downstream migratory barriers to eel sp. and that eel mortality through the structure's working turbines is high (Boubée *et al.,* 1999; McCarthy *et al.,* 2008), signifying the detrimental effects pumping stations have on the two-way migrations of both smaller and larger bodied fish alike.

As is the case for all of the anthropogenic structures assessed in this study, McDowall (1990) reported that the impact of a specific pumping station on a waterway's fish fauna depends on the migratory drive and swimming abilities of a specific species, as well as the physical dimensions of the barrier and its distance from the ocean. The further downstream the barrier is located (Cote *et al.,* 2009) and the closer it is to lowland streams and estuaries (Rolls, 2011), the greater the impact on fish with diadromous life histories. As all of the assessed sub-sites in this study were situated along lowland waterways and each of the studied pumping stations were deemed to act as complete barriers to diadromous fish, this study's drainage structures, as disclaimed by McDowall (1990), have the most important influence on fish community structures as a whole. However, no matter where they are positioned along a waterway, the presence of a pumping station often creates artificial, lentic upstream ecosystems, which can act as "stepping stones" for the continued introduction, establishment and spread of exotic, invasive species (Johnson *et al.,* 2008), like the mosquitofish and goldfish encountered in this study. The subsequent thriving of these species increases the risk of competition and predation for any native, non-migratory species inhabiting a sub-site above a pumping station, affecting the fish at the individual, population, or ecosystem level, leading to contracted ranges and potential localised extinctions (Allibone, 1999; McIntosh *et al.,* 2010).

Complete barrier removal should always be the priority for allowing the upstream passage of migrating fish, especially if the structure is no longer necessary, safe or performing its designated function (Cooney and Kwak, 2013). A study by Rolls (2011) confirmed the importance of dam removal as a significantly greater increase in catchment distribution of the emerald shiner *(Notropis atherinoides),* an economically important bait fish reaching similar average lengths as adult inanga, was reported along the Baraboo River, USA once 4 small dams, roughly equalling the size of the pumping stations in this study were removed. However, due to the high cost and often complex processes associated with dam removal (Cooney and Kwak, 2013), remediation techniques like fishways or trap and transfer programmes are more commonly applied. Up and downstream migration remediations across pumping stations have been widely reported in the literature, though research has primarily focused on eel sp. (McCarthy *et al.,* 2008) and larger bodied salmonids (Mallen-Cooper and Brand, 2007), whilst designs accommodating amphidromous fish with weaker swimming abilities are limited (Boubée *et al.,* 1999; Cooney and Kwak, 2013). A study by Mallen-Cooper and Brand (2007) in Australia assessed the passability of a pumping station with a remediated salmonid fishway for small-bodied native non-salmonids and found that <1% of individuals were able to ascend the fishway due to its high turbulence, high water velocities and unsuitable baffle design, whilst Cooney and Kwak (2013) found that even once amphidromous fish gained upstream access along a Puerto Rican river, downstream drifting larvae still suffered a 42-100% mortality as they approached the pumping station's functioning turbines, signifying the importance of designing a two-way migratory passageway for pumping stations which considers the swimming abilities and ecological requirements of small-bodied amphidromous fish (Jellyman and Harding, 2012) like those encountered in this study.

5.2. The Vulnerability of Differing Life Histories to Anthropogenic Structures

The difference in species richness of upstream, midstream and downstream sub-sites varied according to the type of drainage structure present along each waterway (Fig.4.6). As described above, the species richness was not different between the upstream and the downstream sub-sites along the Mangarau stream where a non-barrier bridge apron was present (Fig.4.6; Table 4.3) and along the Ngaruroro Backwash where a non-barrier culvert was present (Fig.4.6; Table 4.3). Species richness also did not differ between the midstream and the downstream sub-sites along the Muddy Creek where a non-barrier FFFG was present (Fig.4.6; Table 4.3). In contrast, the species richness was lower in upstream sub-sites compared to the richness within downstream sub-sites along the prementioned waterways where a barrier, whether a flood gate or a pumping station was present (Fig.4.6; Table 4.3). There were potential confounding intrinsic effects as fish migrating downstream from freshwater to saltwater environments may have had an effect on the species richness within each sub-site, though this study remains to assess the passability across structures for upstream migrating fish only.

Non-barrier structures allow continued upstream passage for all species, regardless of their life history (Fig.4.5; Table 4.2). The amphidromous inanga and common bully were encountered within both of the upstream and the downstream sub-sites where a bridge apron and a culvert was present, whilst inanga were present within both of the midstream and the downstream sub-sites where a FFFG was present. As amphidromy involves the upstream drifting of small-bodied juveniles with weak swimming abilities (Franklin and Gee, 2019), it is crucial that upstream passage is unrestricted, though the presence of these species within upstream sub-sites indicates that passage is possible at least during a high tide or during high rainfall events where the stream's water levels are at their highest. However, the size of a fish may affect its vulnerability to a structure, where larger bodied or adult fish with stronger swimming abilities may be unaffected by a structure (Birnie-Gauvin *et al*., 2019), whereas small bodied, or juvenile fish with weaker swimming abilities may be unable to migrate upstream (Franklin and Gee, 2019). This may explain the case of the bridge apron, where only one inanga was encountered upstream, suggesting that the structure does not restrict upstream passage to stronger swimming adults, though may act as a barrier to weak swimming juvenile whitebait.

The yellow-eye mullet, shortfin and longfin eels were also able to migrate beyond the non-barriers, indicating that the structures also do not restrict the upstream migrations of catadromous species. Catadromous juvenile fish require free upstream passage to feed and sexually mature, before larger bodied adults migrate back downstream into saltwaters to spawn (Lassalle *et al*., 2008). Although the downstream passage of catadromous fish has not been assessed in this study, it is assumed that adult fish with strong swimming abilities will be able to migrate downstream across the bridge apron, the culvert and the FFFG since the structures did not restrict the upstream passage of weaker swimming juveniles. Although no anadromous or potadromous fish were observed in this study, similar assumptions can be made regarding their vulnerability to non-barrier structures. An anadromous life history involves adult individuals migrating upstream from saltwater to freshwater environments to spawn (Lassalle *et al*., 2008; Birnie-Gauvin *et al*., 2019), whilst potadromous fish migrate fully along freshwaters, though adults require unrestricted passage to spawn upstream (Lassalle *et al*., 2008). Since this study has shown that weaker swimming amphidromous fish are able to drift upstream beyond the non-barrier structures, it is assumed that larger bodied adult fish with anadromous or potadromous life histories will also be unaffected by the structures, and that their upstream migrations can continue unrestrictedly.

Barrier structures, however, restrict the upstream passage of most diadromous fish, regardless of their size, though populations of invasive resident species, or more environmentally tolerant species inhabit the upstream sub-sites above the structures (Fig.4.5; Table 4.2). The amphidromous inanga is absent from all upstream sub-sites above every flood gate and pumping station assessed in this study, though they inhabit almost all of the downstream sub-sites situated below the barrier structures, showing that upstream passage for drifting juvenile inanga is completely restricted. The amphidromous common bully is also absent from most upstream sub-sites, other than along the Ahuriri Station Drain, though it is believed that a sub-population of resident common bully inhabit the stream above the barrier and that the Ahuriri pumping station still acts as a complete barrier to all migrating amphidromous fish.

The effect of the barriers on upstream migrations are not restricted to fish with an amphidromous life history; the catadromous grey and yellow-eye mullet are also absent from all sub-sites above a barrier, though are present within most downstream sub-sites, showing that the flood gates and the pumping stations in this study restrict the upstream migrations of fish with varying life histories. As discussed above, no anadromous or potadromous fish were encountered in this study, though assumptions can still be made regarding their vulnerability to barrier structures. Anadromy and potadromy as life histories involves the upstream migrations of adult fish with stronger swimming abilities (Lassalle *et al.*, 2008; Birnie-Gauvin *et al.*, 2019) than this study's inanga and mullet species which drift or migrate upstream as juveniles (Franklin and Gee, 2019). Therefore, the hydraulic conditions associated with opened flood gates which exceed the swimming capabilities of weak swimming amphidromous and catadromous fish, may not exceed the capabilities of larger bodied anadromous or potadromous fish. However, further research would be required to confirm the vulnerability of fish with these life histories to flood gates and pumping station barriers.

Fish communities within upstream sub-sites situated above a barrier consisted mainly of resident sub-populations of the invasive mosquitofish and goldfish, or the environmentally tolerant New Zealand freshwater shrimp. As discussed in detail in the flood gate and the pumping station sections above, the lack of interspecific competition within sub-sites above a barrier due to the absence of diadromous fish has created ecological niches which invasive species have been able to exploit and thrive within (Johnson *et al.,* 2008; Fuller *et al.*, 2015). Despite acting as physical barriers to all other migrating species, adult catadromous shortfin and longfin eels were encountered within sub-sites situated above flood gates and pumping stations, indicating that the structures do not restrict upstream eel migrations. As also discussed above, eels have much stronger climbing abilities than all other diadromous fish and are able to physically climb from a water channel onto terrestrial land in order to bypass a barrier and to continue their upstream migrations (Jellyman and Harding, 2012). Although not assessed in this study, it has been widely reported in the literature that eel mortality can be as high as 100% across pumping stations, whilst attempting to migrate downstream back to saltwater habitats to spawn (Boubée *et al.,* 1999; McCarthy *et al.,* 2008), emphasising the detrimental impact barrier structures can have on entire sub-populations of a migrating species, even if upstream passage is possible.

5.3. The Effects of Environmental Factors on Fish Abundance

5.3.1. Tidal Influence on Water Chemistries

Although the effect of the tidal cycle on the water temperature, the dissolved oxygen saturation and the specific conductivity has not been assessed in this study, it is important to acknowledge such effects since most of this study's sampled waterways are tidal in nature (Table 4.13). As previously discussed, anthropogenic drainage structures are typically located along, or close to the freshwater-saltwater ecotone boundary zone, where two ecosystems collide (Atrill and Rundle, 2002). The water chemistries within this zone fluctuate according to various natural factors, like the position of the tidal cycle, the time of the day and the height of the tide, along with anthropogenic factors like whether a drainage structure causes habitat dysconnectivity and whether tidal exchange is limited between upstream and downstream sub-sites. The biota of fish within these environmental boundary zones experience a combination of unique challenges and opportunities, which may subsequently influence the abundance and the biodiversity of these species along these sites (Atrill and Rundle, 2002).

It has been widely reported in the literature that the temperature and the dissolved oxygen saturation of an estuarine zone, i.e., areas along either control waterways or along downstream sub-sites where a structure was present in this study, varies temporally, according to the time of the day (Nelson *et al.,* 1994; Fortune and Mauraud, 2015). Temperatures increase during sunlight hours as the amount of solar radiation that the water channel is exposed to increases, whilst the dissolved oxygen saturations decreases (Nelson *et al.,* 1994). The highest temperatures are associated with the lowest tides and the lowest temperatures with the highest tides (Nelson *et al.,* 1994; Fortune and Mauraud, 2015). As the tide recedes, the amount of nutrient mixing is reduced, and the dissolved oxygen saturation levels subsequently decreases. The specific conductivity of a waterway is highest during a high tide where advancing flood tides with higher SPC measures collide with the freshwater ecotones which are typically associated with lower SPCs (Miller *et al.,* 1988; Fortune and Mauraud, 2015). As the tide recedes, the specific conductivity of the waterway decreases (Miller *et al.,* 1988).

All of the tidal waterways used in this study experienced a low tide within the morning of the sampled day, with the exception of the lower Taipo stream, County Drain and Plantation Drain which all experienced morning high tides (Table 4.13), although the time of the lowest and the highest tides varied for each site. The effects of the tide on the environmental factors were not assessed in this study, though efforts were made during the sampling process of the methodology to sample fish abundances and measure water chemistries at similar positions of the tidal cycle to ensure consistency within the results, although this was not always possible. Therefore, the following sections discuss the effect of temperature, dissolved oxygen saturation and specific conductivity on the abundance of fish, without considering any tidal influence.

5.3.2. Temperature and Dissolved Oxygen

Whilst considering the physical effects of each of the assessed drainage structures, this study found that the predicted abundance of inanga (Fig.4.11; Table 4.12) and the total abundance of all species (Fig.4.17; Table 4.12) significantly decreased as the level of dissolved oxygen saturation within the water channel increased for all treatments, though the rate of increase differed between control, bridge apron, culvert, flood gate, pumping station and PS/ FFFG sites, depending on their barrier

status. The predicted abundance of inanga (Fig.4.10. Table 4.12) and the predicted total abundance of all species (Fig.4.16; Table 4.12) increased as the water temperature increased for all treatments, though the rise in the total abundance of all species was not significant.

The findings of this study were not as expected and are in contrast to those commonly reported in the literature which generally find that a greater abundance of native, migratory species like inanga, inhabit high quality environments with lower water temperatures and high dissolved oxygen (DO) saturations. Studies by Richardson *et al.* (1994) and Dean and Richardson (1999) offer a potential explanation for the differences observed between the findings of this and previous studies. The largest inanga populations encountered in this study, which influenced the overall trends observed within the data, were along a control and a non-barrier sub-site, each with considerably lower dissolved oxygen saturations compared with the other assessed waterways. The water qualities of these sites however remain within, or close to the thresholds that inanga are able to tolerate, which Dean and Richardson (1999) reported to be approximately 36% DO saturation. The temperature readings of the two highly populated waterways lay between 18.05-21.35°C, close to the preferred adult inanga preference of 18.1°C and far from the lethal range of 30.5-35.4°C as reported by Richardson *et al.* (1994). Although juvenile and adult inanga are highly impacted by physical migratory barriers as discovered in this study, they remain tolerant of fluctuating environmental conditions and are able to inhabit 'degraded' waterways with higher temperatures and lower dissolved oxygen saturations than other, small-bodied diadromous fish (Fake, 2018).

Anthropogenic drainage structures acting as full or partial barriers to migrating fish often also alter the hydrological and thermal processes of a waterway, leading to the modification or loss of essential aquatic habitats (Birnie-Gauvin *et al.,* 2019) and the potential rise of environmental barriers, although such effects were not recorded in this study. Therefore, it is essential to address the environmental impact of drainage structures so that habitat mitigations can be made simultaneously with the physical fish passage remediations previously discussed in this study.

As described in the literature and supported by the water qualities of sites assessed in this study, control waterways, or those without a structure present, and bridge apron and culvert waterways, with structures categorised as partial, or 'non-barriers' to migrating fish, do not act as considerable environmental barriers, whereas flood gates and pumping stations, structures categorised as full migratory barriers, cause extreme upstream habitat modifications with higher water temperatures and lower dissolved oxygen saturations than streams where a barrier is not present. Flood gates and pumping stations limit the tidal exchange between up and downstream sub-sites, altering the flow regime of habitats beyond the barriers, reducing sediment transport along the water channel and encouraging the accumulation of excess nutrients upstream (Van Puijenbroek *et al.,* 2019). The lack of mixing between sub-sites has led to the degradation of the overall quality of fish habitat above barriers in this study with high temperature and low dissolved oxygen saturations which exceed the tolerances of sensitive species and are generally considered too lethal for native New Zealand fish communities to inhabit (Bocker, 2015). Jellyman and Harding (2012) reported that barriers, like those assessed in this study often alter an upstream ecosystem from lotic to one which is degraded and lentic, leading to a fish community shift consisting primarily of lentic specialising species (Gao *et al.,* 2010). However, the lack of these specialists in New Zealand's fish fauna creates a vacant niche, which invasive fish, like mosquitofish and goldfish have successfully filled; an observation consistent to the findings of this study and studies around the world (Han *et al.,* 2008).

A study by Joy (2015) found that the lack of tidal exchange between sub-sites when an environmental barrier is present, and the subsequent accumulation of excess nutrients upstream often leads to large daily fluctuations in temperature and dissolved oxygen saturations. During

periods of stagnation, hypoxic conditions form which can extend over 100m upstream (Kroon and Ansell, 2006), causing increasing water temperatures and further degrading crucial habitats for potadromous and resident riverine fish (Van Puijenbroek *et al.,* 2019). Water temperature and dissolved oxygen measures are often inversely proportional to one another, where a high water temperature is able to hold fewer oxygen molecules than cooler waters and is therefore associated with a low dissolved oxygen saturation and vice versa (Kroon and Ansell, 2006), as observed in this study.

Numerous studies within the literature relate high water temperatures with the absence or simplified nature of a stream's riparian vegetation (Joy and Death, 2001; Franklin and Hodges, 2015; Pelicice *et al.,* 2015). The lack of native, overhanging vegetation along the riparian strips of many of this study's upstream sub-sites (Table 2.2) has limited the amount of shading cast across each waterway and has exposed each stream to excessive levels of sunlight (Roy *et al*., 2003). Miserendino *et al.* (2011) found that significantly higher shading was cast across streams which ran through native, non-managed forests than across managed or urban waterways and that significantly lower water temperatures were recorded within the most shaded regions. Recent studies have shown that increased shading cast by riparian vegetation can have the potential to reduce stream temperatures by up to $10^{o}C$ and limit the daily and seasonal variation which commonly occurs along lowland streams and estuaries (Mueller *et al*., 2011; Gibson *et al*., 2017), so that water conditions remain stable and within the relatively low thresholds for New Zealand's migratory fish (Richardson *et al.,* 1994).

Studies by Roy *et al.* (2003), Miserendino *et al.* (2011) and Joy (2015) reported that degraded or simplified riparian strips, whether through anthropogenic clearing or by livestock overgrazing, leads to an increase in the trapping and retention of fine sediment and the reduction of suitable instream egg laying or adult emergence sites, like woody debris and leaf litter, or larger substrates like boulders and cobbles. An increase in nutrient accumulation can occur, which may result in the excessive overgrowth of algae and aquatic plants, including the invasive and ecologically harmful hornwort *(Ceratophyllum demersum),* a macrophyte known to displace native, submerged vegetation and one found within this study's sub-sites where low dissolved oxygen and high temperatures were recorded.

5.3.3. Specific Conductivity

Whilst considering the physical effect of the drainage structures in this study, the predicted abundance of inanga marginally increased as the specific conductivity increased across all treatments, although the differences were not significant (Fig.4.12; Table 4.12). The predicted total abundance of all species also increased as the SPC increased across all treatments in this study and the differences were significant (Fig.4.18; Table 4.12). The differences in the predicted abundance of inanga and the predicted total abundance of all species between treatments were minor and were irrespective of their barrier status.

Specific conductivity (SPC), a measure most commonly used to assess water quality and changes in an ecosystem, is directly related to the concentration of ions produced as salts and inorganic materials dissolve, where measures ranging between 150-500 µS/cm indicate a healthy, good quality freshwater habitat, whilst sea water environments can reach SPC levels of up to 50,000 µS/cm (Gordos *et al.,* 2007). The most significant differences in SPC in this study were encountered between the upstream and the downstream sub-sites along waterways where a pumping station

was present, findings consistent with those by Russell *et al.* (1998) who reported that an 8m high barrage along the Tawe estuary, South Wales, UK, limited the tidal flushing and subsequent mixing of upstream, lentic freshwater lakes with low SPCs and downstream, lotic saltwaters with higher SPC measures. As pumping stations and barrages (Russell *et al.,* 1998) both prevent the upstream physical passage of diadromous fish, as previously reported in this study, individuals are limited to, and most abundant in downstream habitats. Therefore, a greater abundance of fish inhabiting ecosystems with higher SPC levels may in fact be due to the inability of a fish to physically navigate beyond a barrier into upstream freshwater habitats, rather than the actual preference of inhabiting saltwaters with higher SPC levels (Boys *et al.,* 2012; Bocker, 2015).

Although categorised as a complete physical barrier to migrating fish, SPC levels above and below flood gates in this study only differed to a small extent, with slightly greater SPCs encountered in downstream sub-sites, findings consistent with those by Pollard and Hannan (1994). Their study also reported that only slight differences in salinity measures (associated with SPC) were observed between sites with and without a flood gate along the Clarence River, Australia, and that the structures were "generally ineffective" in preventing the mixing of fresh and saltwaters. The relationship between increasing fish abundance and rising SPC levels in this study was mirrored by Pollard and Hannan (1994) who further concluded that this relationship was directly correlated to the distance the waterways were from the open ocean. Diadromous fish abundances and species richness were often greatest in downstream sites closer to the sea in their study where SPC levels were generally at their highest, particularly when a physical barrier was preventing upstream access (Miserendino *et al.,* 2011). However, a study by Obolewskt *et al.* (2018) reported contrasting results; they found that a significant decline in the SPC levels of upstream habitats following a flood gate installation along the Jamno lagoon, Poland, led to the transformation of a brackish ecosystem into one which was fully freshwater, impacting the abundance and biodiversity of local upstream fish and macroinvertebrate communities. The contrasting findings between this study and that by Pollard and Hannan (1994) and the Obolewskt *et al.* (2018) report highlights the differing effects a drainage structure can have on a specific habitat and emphasises the need to assess, remove or retrofit barriers on an individual basis.

If a structure is not present, or if upstream access is not physically restricted, as is the case for the control, the bridge apron and the culvert sites in this study, the level of SPC does not have a considerable effect on the abundance of inanga. Inanga were encountered along control sites varying in distance from the open ocean, including in freshwater ecosystems with SPC levels of <400 µS/cm, and within downstream saltwater ecosystems with SPC levels >22,000 µS/cm, though habitats close to a stream's freshwater-saltwater transition zone are typically the preferred location for mass inanga inhabitation and spawning (Hicks *et al.,* 2010). Due to the close proximity of the up and the downstream sub-sites along the waterways where a bridge apron or a culvert was present, and the fact that neither structure acted as a full physical barrier as previously discussed in this study, SPC levels did not significantly differ and therefore did not have an effect on the fish communities encountered along these waterways. Although fish ramp installation onto these structures may aid the upstream passage of migrating fish, it is unlikely that these retrofits will lead to significant alterations in the water chemistries of either sub-site.

5.4. Barrier Mitigation and Site Remediation

By improving the connectivity and the tidal flow between upstream and downstream sub-sites where a physical structure is present, environmental barriers are often also mitigated. If a barrier is removed, or a fish ramp or a 'fish-friendly' mechanism is installed onto a structure as previously discussed in this study, the 'River Continuum Concept' (Vannote *et al.,* 1980) predicts that sediment transport, tidal flushing and dissolved oxygen mixing between sub-sites will increase and the upstream habitat will at least in part, naturally remediate (Joy and Death, 2001). The positive environmental impacts associated with reconnecting previously isolated habitats through structure remediation have received some attention in the literature. Studies by Roni *et al*. (2008) and Boys *et al.* (2012) described the improvements in dissolved oxygen and water temperatures encountered upstream once a fish ramp or a FFFG retrofit was installed onto a drainage structure and the subsequent rise in diadromous fish abundance, biodiversity and species richness within upstream sub-sites that followed. However, Bocker (2015) and Franklin and Hodges (2015) found slightly contrasting results, with findings consistent to those in this study. Bocker (2015) reported that no significant decrease in water temperatures occurred following the installation of a FFFG, despite a rise in dissolved oxygen saturations along 2/3 of their sampled waterways, whilst Franklin and Hodges (2015) observed a decline in dissolved oxygen after retrofitting a flood gate. Their study went on to explain that the increase in tidal flushing between sub-sites post FFFG installation remobilised deposited nutrients within the upstream water channel, leading to increases in oxygen demand and lower overall dissolved oxygen saturations. This may offer an additional explanation as to why a large population of inanga were encountered above a FFFG in this study (Table 4.12), despite the sub-site's low dissolved oxygen saturation. The reduction in the upstream water quality however may only be temporary and may settle once excess nutrient and sediment build up is flushed downstream over time (Franklin and Hodges, 2015).

Studies reporting the alterations in SPC levels following pumping station remediation are sparse within the literature, though Boys *et al.* (2012) and Bocker (2015) show that increases in upstream conductivity occurred following the installation of a 'fish-friendly' mechanism onto an existing flood gate. As the retrofit delays the closing of the gate, a greater influx of saltwater occurs and a gradual transition zone between fresh and saltwaters is restored (Russel *et al.,* 1998), leading to increased SPC levels and higher upstream fish abundance and biodiversity. Both studies however concur that the rise in biodiversity was solely due to improved physical passage above the flood gate, rather than the increase in SPC within the upstream habitat. Further assessments are required to ensure that no negative environmental impacts arise as a result of the FFFG installation. As discussed by Gordos *et al*. (2007) and Johnson *et al*. (2008), if the reopening of flood gates is not carefully managed, issues regarding the overtopping of saltwater into adjacent agricultural or pastoral lands or increased lateral seepage into shallow groundwater may occur, affecting lowland agricultural industries and human infrastructure, highlighting the potential trade-offs associated with retrofit installations and the need to assess a habitat post-remediation.

Additional human induced riparian remediations may be required along the most degraded waterways to further improve and rehabilitate ecosystems and to reverse the negative environmental impacts caused by anthropogenic barriers. Previous studies by Roni *et al.* (2002; 2008) discussed the ecological benefits of planting large, overhanging native trees, shrubs, tussocks and grasses along each bank to cast shading across the width of a stream, in order to decrease water temperatures, reduce the abundance of excessive macrophytes and subsequently increase the dissolved oxygen saturations (Bocker, 2015). Renewed riparian vegetation would also provide additional suitable fish and invertebrate spawning, refuge and feeding habitat (McRae *et al*., 2016)

and act as a waterway's natural buffer against bankside erosion, reducing the volume of fine sediment released into the water channel and leading to a rise in habitat complexity with a diverse structure of pools and riffles (Miserendino *et al.,* 2011). Further research would be required to assess the extent of how degraded a habitat above a barrier is, to determine whether additional remediation of the riparian strips is necessary, or whether the implementation of a drainage structure retrofit will suffice.

5.5. Study Limitations

As with most studies assessing quantitative data, limitations arose during the data collection and analysis in this study. Although five separate waterways were assessed as control sites, and six for pumping stations, a flood gate was only present along three waterways and a FFFG, a bridge apron and a culvert were only present along one waterway each. It is important to acknowledge the limitations and potential impacts on the results when a differing number of study sites are used for the analysis of each treatment. Naturally, control and pumping station sites had the largest sample sizes within the study, ensuring that the statistical results of these treatments were the most precise and representative. However, single-site treatments like the bridge apron assessments had considerably smaller sample sizes, yielding less precise statistical results with wide confidence intervals. In order to improve the reliability of these results in future studies, an equally large number of study sites should be assessed for all treatments which would consider the variation between sites and improve the reliability of statistical analyses. In saying this, the nature of this study where sample sizes of zero are common, i.e., as observed within sub-sites above barriers in the inanga abundance assessments, 'less precise' statistical results with wide confidence intervals may be unavoidable.

As previously discussed by Pollard and Hannan (1994), the distance a study site is situated from the open ocean affects the abundance of diadromous fish or the richness of a fish community within that waterway. Although all sites in this study were located along lowland waterways within the same county, sites were chosen based on their presence or absence of an anthropogenic drainage structure and the knowledge, or assumptions that inanga inhabited those sites, with little attention given to the geographical distance the sites lie from the open ocean. These varying distances may have acted as an additional factor affecting the abundance of inanga and other migrating fish, other than the passability of the structure or the environmental qualities analysed in this study. Future studies should include the distance of the waterway from the open ocean as a factor in the statistical analyses, enabling further assessments looking at the distribution of inanga along rivers, and whether their distribution is impacted by the passability of certain types of anthropogenic drainage structure.

5.6. Further Research

The results of this study confirm that the passability of various small-scale anthropogenic drainage structures for small-bodied diadromous fish with weak swimming and climbing abilities varies according to the treatment, with bridge aprons and culverts acting as partial, or non-barriers, and flood gates and pumping stations acting as full, or complete barriers. Although offering insightful knowledge for site-specific structure retrofitting and habitat remediations to improve local native fish communities, it must be emphasised that these findings can also be directly compared to previous or future international studies to assess the passability of similar sized drainage structures on *Galaxias maculatus* within lowland waterways which exhibit similar local habitat and environmental features, as previously explained in this chapter.

Widely distributed across the Southern Hemisphere, *Galaxias maculatus* are vulnerable to habitat fragmentation, as discovered in this study, though assessments on the quantity and the dispersion of small-scale anthropogenic drainage structures in many areas beyond New Zealand, like across South America are limited within the literature. Further research would explore the impact of these structures on the migrations of the South American populations of *Galaxias maculatus,* with the findings being directly comparable with those from this study if similar sized and designed structures were assessed. Retrofitting installations like the fish ramps and the FFFGs suggested in this study could then be trialled on structures across South America with the aim of improving local, ecosystem-wide and continent-wide native fish populations. A potential limitation of this further research is that all environmental conditions will need to be consistent between studies in order to directly compare the effects of the South American structures to those in this study and to ensure that the structures are the sole cause of a change in fish abundance or biodiversity. Although this is not always possible due to differing climatic conditions between countries, the effect of environmental factors on the South American *Galaxias maculatus* may also offer an interesting approach for future research.

The results of this study and any subsequent additional further research could branch beyond the scope of inanga across the Southern Hemisphere and can be applied to various other weak swimming amphidromous fish across the Northern Hemisphere, including within the UK. In contrast to the South American continent, the abundance and the distribution of small- and large-scale anthropogenic drainage structures across the UK has been well documented. Jones *et al.* (2019) described that there is >1 structure for every 1.5km of stream across the country, whilst only 1% of Welsh, English or Scottish waterways are free of an artificial barrier entirely. These figures, coupled with the findings of this study, signifies the colossal amount of stream fragmentation and habitat loss or degradation which exists along the UK's waterways and the potentially detrimental effect these structures have on local, native fish communities. Despite this, and mirrored across the Northern Hemisphere, many British studies have focused on the effect of large-scale hydropower dams (Environment Agency, 2018) or the effect of barriers on large-bodied salmonid sp. only (Buddendorf *et al.,* 2019). Hence, future research could adopt a similar methodology as the one used in this study, applying it to assess the passability of small-scale drainage structures for small-bodied migrating fish in the UK.

Due to the higher annual levels of rainfall exhibited across the UK compared with those in Hawke's Bay, New Zealand (Dowling, 1999), additional environmental barriers may exist across certain small-scale drainage structures. Further research could show the effect of increased water velocity on the passability of bridge aprons, culverts or flood gates for fish with similar weak swimming abilities in the UK, with results being comparable to those in this study where rainfall levels were generally

much lower. Assumptions could then be made on how the structures in this study may further affect the upstream migrations of inanga if a high rainfall event was to occur in Hawke's Bay.

Using the fairly straight forward, cheap and reproducible methodology of this study, the future research suggested here could be used to interpret the impacts of anthropogenic drainage structures on the dynamics of a population along a stream, and whether these impacts materially affect the size, the resilience and the stability of local fish sub-populations. If the impacts of a barrier are extensive, results could inform site-specific remediations of local habitats and could encourage conservation efforts to protect threatened fish populations. Once either a specific structure, or a certain type of structure is proven to act as a barrier, whether partial or full as in this study, appropriate remediations can be implemented with the aim of allowing full upstream access for migrating fish, leading to an increase in local fish populations and the enhancement and maintenance of healthy ecosystems.

Chapter 6
Conclusions

The majority of the literature studying the effects of human-induced waterway fragmentation caused by anthropogenic drainage structures focuses on the impacts of either large-scale barriers like dams on Northern Hemisphere fish species, or the effect of these barriers on large-bodied anadromous salmonid sp. with strong swimming and climbing abilities. Despite the wide distribution of *Galaxias maculatus* across the Southern Hemisphere, and their economic, recreational and cultural significance across New Zealand, the effect of small-scale drainage structures on the upstream migrations of inanga is not well understood. With weak swimming and climbing abilities, the amphidromous inanga are highly susceptible to physical barriers within lowland, coastal waterways as they attempt to migrate upstream to sexually mature and spawn, before eventually drifting back downstream from freshwater to saltwater habitats as newly hatched larvae. The aims of this study were to assess the passability of various small-scale drainage structures on lowland fish communities in Hawke's Bay, New Zealand where inanga were known or assumed to be present and the effect, if any, of specific water chemistries on the abundances of the same fish populations.

This study found that the barrier status of each treatment along a lowland waterway does affect the abundance of inanga and the total abundance of all species to different degrees, as bridge aprons and culverts act as partial barriers only, allowing upstream access for adult or large bodied fish during periods of a high tide, whilst flood gates and pumping stations act as full and complete barriers to migrating fish. Further assessments above and below various drainage structures found that the biodiversity differed between sub-sites for all treatments, apart from where a bridge apron was present. Where the total abundance of all species was higher upstream of a structure deemed to act as a barrier in this study, it was discovered that resident populations of the invasive mosquitofish and goldfish, and more environmentally tolerant species like the New Zealand freshwater shrimp and the common bully accounted for almost all of the individuals inhabiting upstream sub-sites, whereas native fish with amphidromous or catadromous life cycles, like inanga and grey and yellow-eye mullet were encountered downstream only.

The abundance of inanga and the total abundance of all species were found to increase with increasing temperature and decreasing dissolved oxygen saturations, whilst increasing specific conductivity led to an increase in the total abundance of all species only. The high abundance of inanga in habitats with water qualities typically described as 'degraded' in the literature signifies the important role that even 'degraded' waterways play as crucial feeding, refuge and spawning grounds and highlights the need to reconnect all habitats fragmented by anthropogenic barriers through structure removal or retrofit installation.

The numbers of small-scale drainage structures, like those assessed in this study, often exceed large-scale dams by up to a thousand times globally, therefore the findings of this research directly contributes to the wide-scale knowledge of how anthropogenic barriers affect the migrations of native, diadromous fish, with a particular focus given to the amphidromous inanga. Appropriate remediations can then be suggested, like the fish ramps and the FFFGs previously discussed, with additional assessments conducted to ensure that these retrofits successfully aid upstream fish

migrations. A previously installed FFFG analysed in this study showed that a significantly higher abundance of inanga and a significantly higher total abundance of all species were encountered above the FFFG, whilst biodiversity measures were significantly different between sub-sites, signifying that the retrofitted structure acts as a complete non-barrier and allows full upstream passage into suitable spawning habitat. As inanga are considered weak swimmers, the fact that they were able to overcome a FFFG infers that the passage of stronger swimming and climbing fish is also possible beyond the structure. Global authorities at the local and national levels can use the widely applicable findings of this study at comparable small-scale drainage structures when deciding which retrofit solution to install in order to enable the most efficient passage of similar fish with similar swimming abilities. The environmental analyses and the site-specific habitat assessments conducted in this study can also be used to collaborate with local landowners, ecologists and cultural groups, as in the case of the Māori in New Zealand, to determine the most efficient ways to restore a habitat to the natural state it once was prior to structure construction, whilst satisfying the requirements of all of those involved.

The findings of this study and the subsequent remediation of small-scale anthropogenic drainage structures will lead to increased native fish abundances, a rise in biodiversity and improved upstream habitat qualities for many lowland waterways, contributing to the maintenance of healthy, functional ecosystems and the lowering of conservation statuses for ecologically, recreationally and culturally important fish species worldwide.

Chapter 7
Appendices

7.1. Appendix A – Data Sampling and Processing

7.1.1. Habitat Assessments

Habitat parameter	Condition category										SCORE
1. Deposited sediment	The percentage of the stream bed covered by fine sediment.										
	0	5	10	15	20	30	40	50	60	≥ 75	
SCORE	10	9	8	7	6	5	4	3	2	1	
2. Invertebrate habitat diversity	The number of different substrate types such as boulders, cobbles, gravel, sand, wood, leaves, root mats, macrophytes, periphyton. Presence of interstitial space score higher.										
	≥ 5	5	5	4	4	3	3	2	2	1	
SCORE	10	9	8	7	6	5	4	3	2	1	
3. Invertebrate habitat abundance	The percentage of substrate favourable for EPT colonisation, for example flowing water over gravel-cobbles clear of filamentous algae/macrophytes.										
	95	75	70	60	50	40	30	25	15	5	
SCORE	10	9	8	7	6	5	4	3	2	1	
4. Fish cover diversity	The number of different substrate types such as woody debris, root mats, undercut banks, overhanging/encroaching vegetation, macrophytes, boulders, cobbles. Presence of substrates providing spatial complexity score higher.										
	≥ 5	5	5	4	4	3	3	2	2	1	
SCORE	10	9	8	7	6	5	4	3	2	1	
5. Fish cover abundance	The percentage of fish cover available.										
	95	75	60	50	40	30	20	10	5	0	
SCORE	10	9	8	7	6	5	4	3	2	1	
6. Hydraulic heterogeneity	The number of of hydraulic components such as pool, riffle, fast run, slow run, rapid, cascade/waterfall, turbulance, backwater. Presence of deep pools score higher.										
	≥ 5	5	4	4	3	3	2	2	2	1	
SCORE	10	9	8	7	6	5	4	3	2	1	
7. Bank erosion	The percentage of the stream bank recently/actively eroding due to scouring at the water line, slumping of the bank or stock pugging.										
Left bank	0	≤ 5	5	15	25	35	50	65	75	> 75	
Right bank	0	≤ 5	5	15	25	35	50	65	75	> 75	
SCORE	10	9	8	7	6	5	4	3	2	1	
8. Bank vegetation	The maturity, diversity and naturalness of bank vegetation.										
Left bank AND Right bank	Mature native trees with diverse and intact understorey		Regenerating native or flaxes/sedges/tussock > dense exotic			Mature shrubs, sparse tree cover > young exotic, long grass			Heavily grazed or mown grass > bare/impervious ground.		
SCORE	10	9	8	7	6	5	4	3	2	1	
9. Riparian width	The width (m) of the riparian buffer constrained by vegetation, fence or other structure(s).										
Left bank	≥ 30	15	10	7	5	4	3	2	1	0	
Right bank	≥ 30	15	10	7	5	4	3	2	1	0	
SCORE	10	9	8	7	6	5	4	3	2	1	
10. Riparian shade	The percentage of shading of the stream bed throughout the day due to vegetation, banks or other structure(s).										
	≥ 90	80	70	60	50	40	25	15	10	≤ 5	
SCORE	10	9	8	7	6	5	4	3	2	1	
TOTAL	(Sum of parameters 1-10)										

Figure 7.1. The National Rapid Habitat Assessment Protocol Development for Streams and Rivers form used to describe and assess the habitat of each sub-site sampled across this study, sourced from the Cawthron Institute (Clapcott, 2015).

7.1.2. Species Abundance

Statistical analyses were conducted using the 'DHARMa', 'performance' or 'countreg' packages in RStudio, version 3.5.1.

7.1.2.1. Inanga Abundance Vs Barrier Status
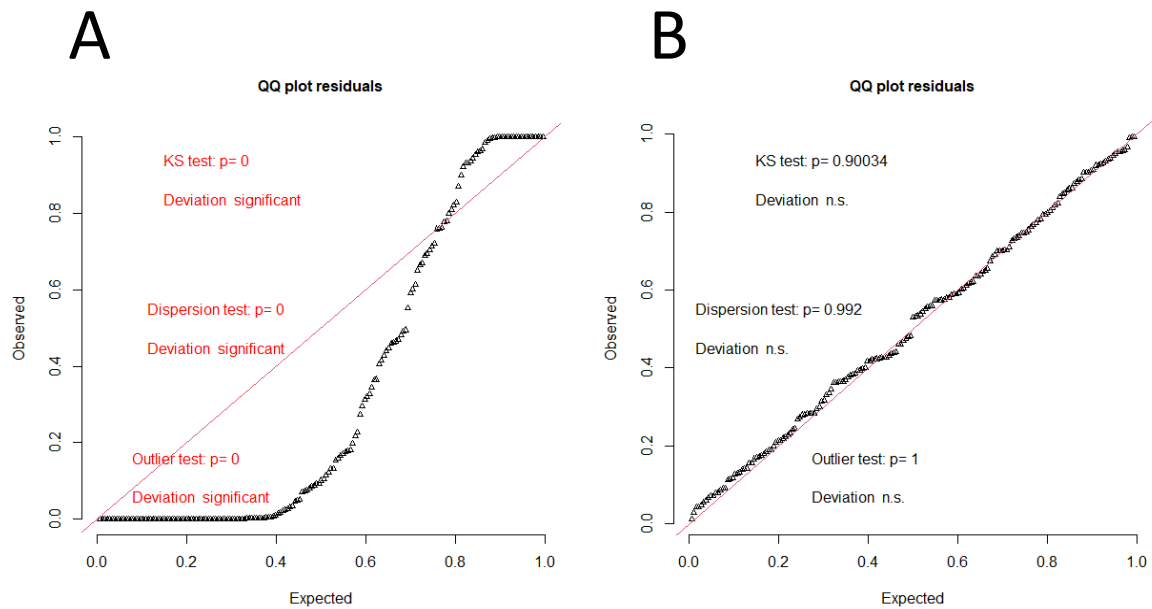
Figure 7.2. Comparison of Poisson and negative binomial distributions of 'Inanga Abundance Vs Barrier Status' dataset.

Quantile residual QQ plots with outlier, dispersion and KS tests for the 'Inanga Abundance Vs Barrier Status' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B:** Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'Inanga Abundance Vs Barrier Status' dataset. Statistical models were compared using the 'DHARMa' package in RStudio, version 3.5.1.
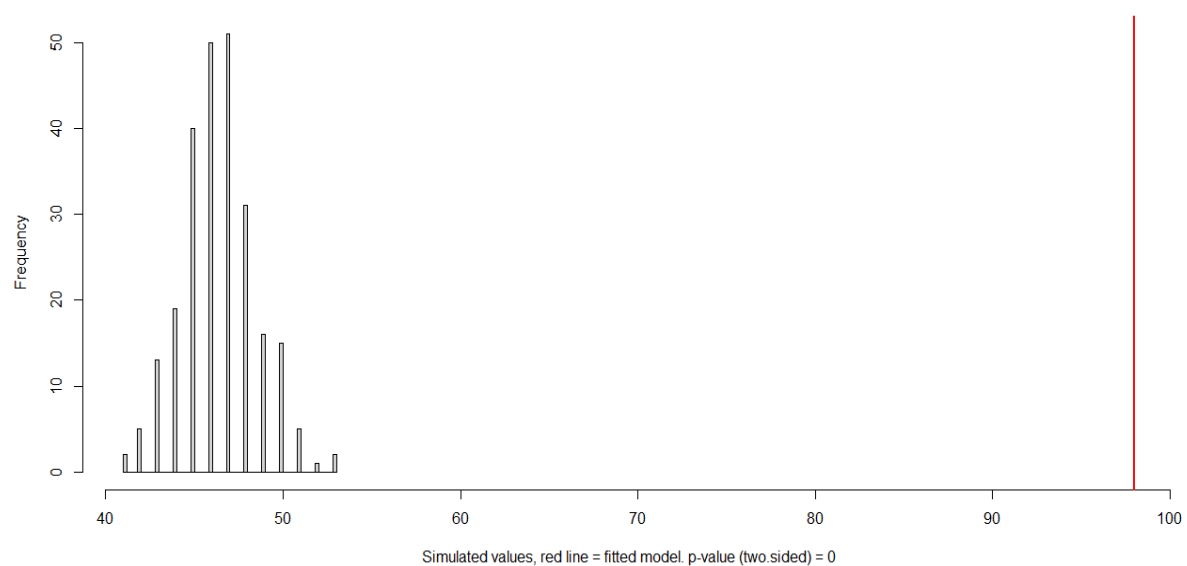
Figure 7.3. Zero-inflation plot for the 'Inanga Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.

Poisson

Figure 7.4. Rootogram showing the Poisson distribution of the 'Inanga Abundance Vs Barrier Status' dataset. Strong over and underpredictions of most counts indicate that overdispersion exists and excess zeros are unaccounted for.

Simulated values, red line = fitted model. p-value (two.sided) = 1

Figure 7.5. Zero-inflation plot for the 'Inanga Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
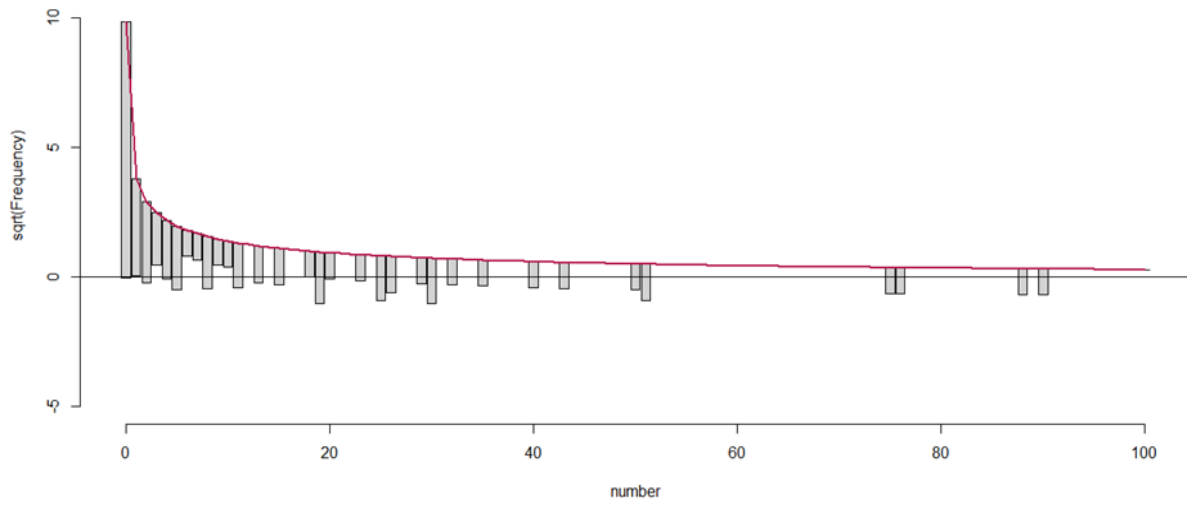
Figure 7.6. Rootogram showing the Neg Bin distribution of the 'Inanga Abundance Vs Barrier Status' dataset. Under and over predictions from expected counts are small and the zero-count is well fitted, indicating a suitably fit model.

7.1.2.2. Inanga Abundance Vs Treatment

Figure 7.7. Comparison of Poisson and negative binomial distributions of 'Inanga Abundance Vs Treatment' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLM for the 'Inanga Abundance Vs Treatment' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B**: Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'Inanga Abundance Vs Treatment' dataset.

Figure 7.8. Zero-inflation plot for the 'Inanga Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.

Poisson

Figure 7.9. Rootogram showing the Poisson distribution of the 'Inanga Abundance Vs Treatment' dataset. Strong over and underpredictions of most counts indicate that overdispersion exists and excess zeros are unaccounted for.

Figure 7.10. Zero-inflation plot for the 'Inanga Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
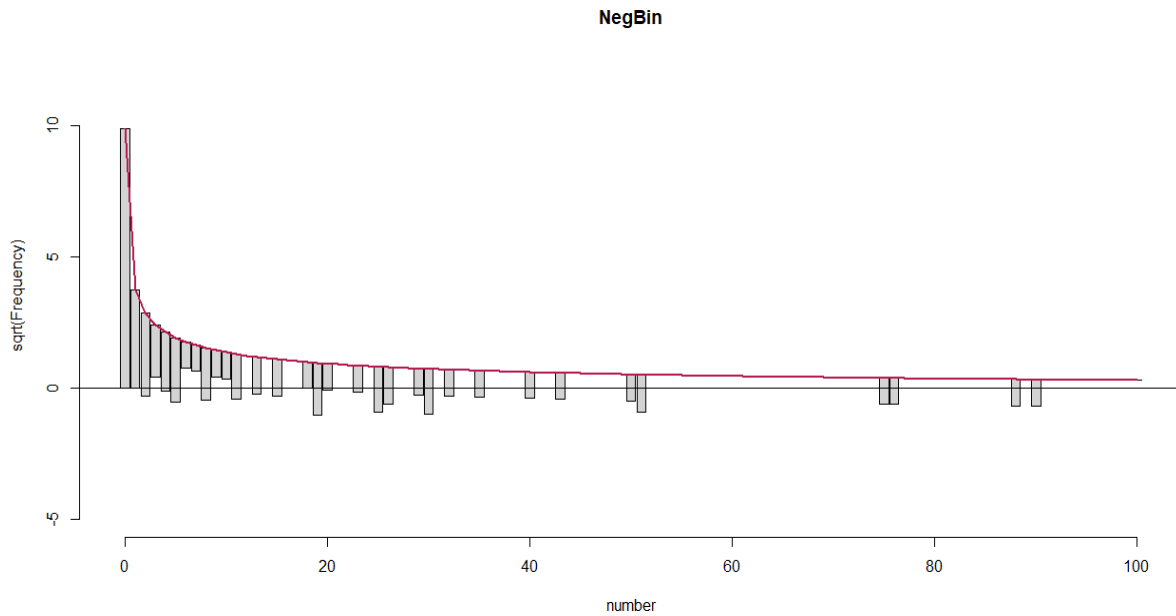
NegBin

Figure 7.11. Rootogram showing the Neg Bin distribution of the 'Inanga Abundance Vs Treatment' dataset. Under and over predictions from expected counts are small and the zero-count is well fitted, indicating a suitably fit model.

7.1.2.3. All Species Total Abundance Vs Barrier Status

Figure 7.12. Comparison of Poisson and negative binomial distributions of 'All Species Total Abundance Vs Barrier Status' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLM for the 'All Species Total Abundance Vs Barrier Status' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B:** Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'All Species Total Abundance Vs Barrier Status' dataset.

Figure 7.13. Zero-inflation plot for the 'All Species Total Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
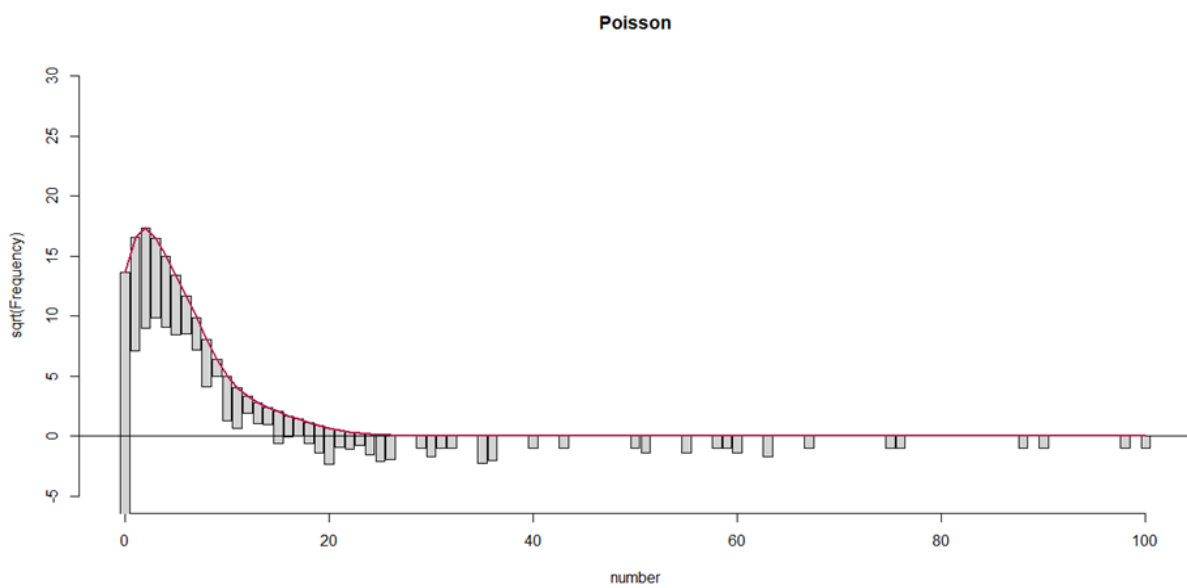
Figure 7.14. Rootogram showing the Poisson distribution of the 'All Species Total Abundance Vs Barrier Status' dataset. Strong over and underpredictions of most counts indicate that overdispersion exists and excess zeros are unaccounted for.

Simulated values, red line = fitted model. p-value (two.sided) = 0.864
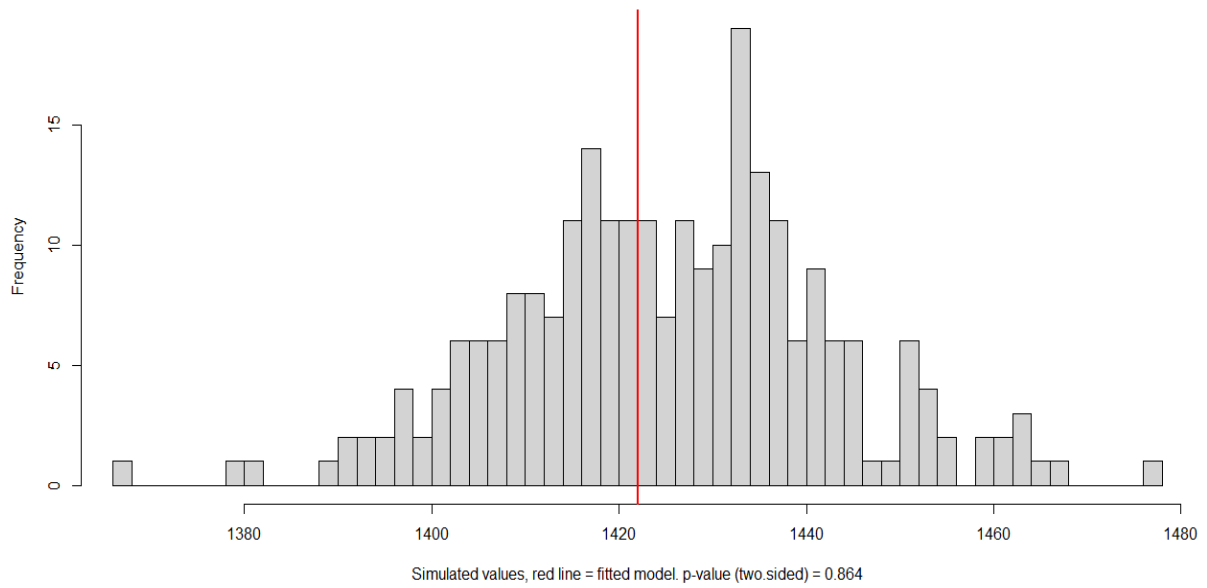
Figure 7.15. Zero-inflation plot for the 'All Species Total Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
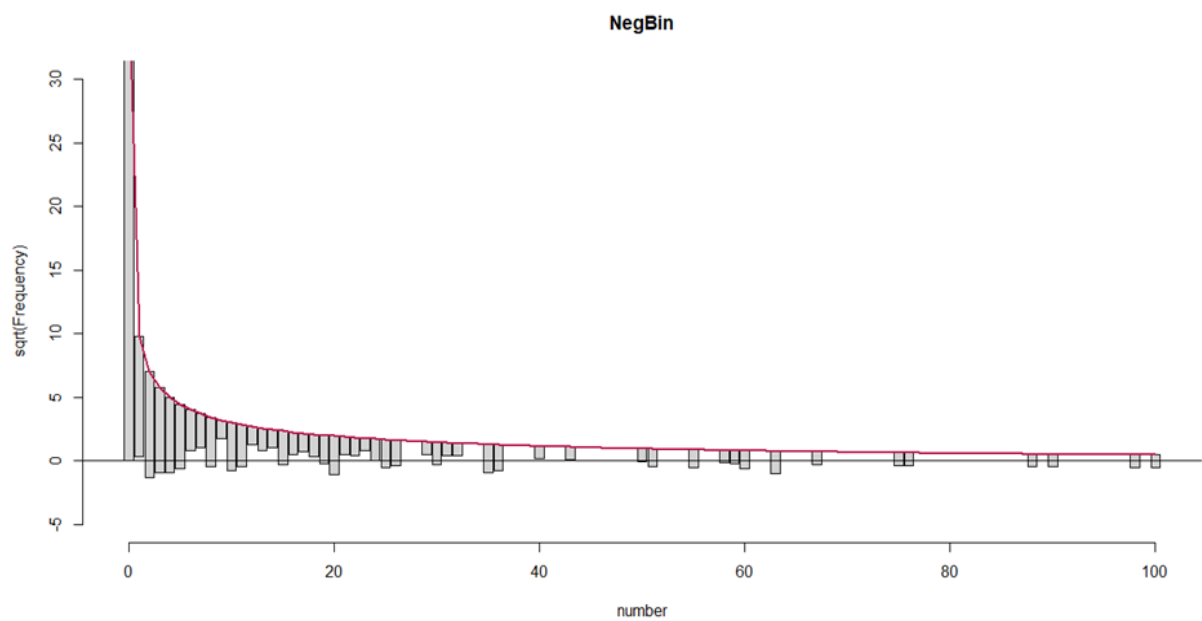
Figure 7.16. Rootogram showing the Neg Bin distribution of the 'All Species Total Abundance Vs Barrier Status' dataset. Under and over predictions from expected counts are small indicating a suitably fit model, though the zero bin is large and may need to be accounted for using a zero-inflated model.

7.1.2.4. All Species Total Abundance Vs Treatment
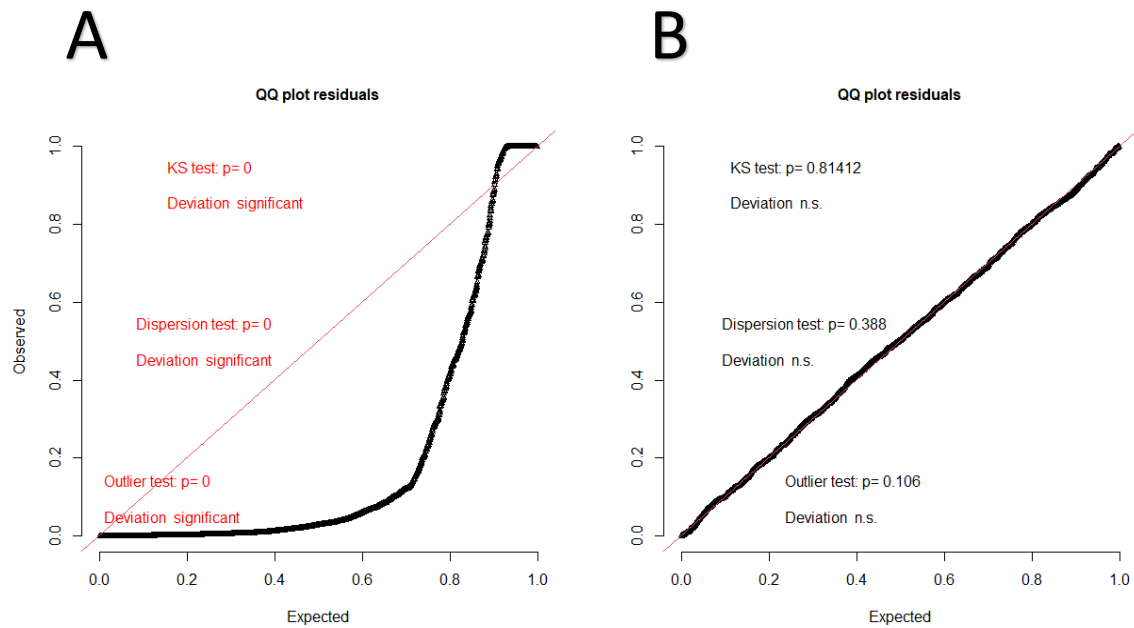
Figure 7.17. Comparison of Poisson and negative binomial distributions of 'All Species Total Abundance Vs Treatment' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLM for the 'All Species Total Abundance Vs Treatment' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B:** Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'All Species Total Abundance Vs Treatment' dataset.

Figure 7.18. Zero-inflation plot for the 'All Species Total Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
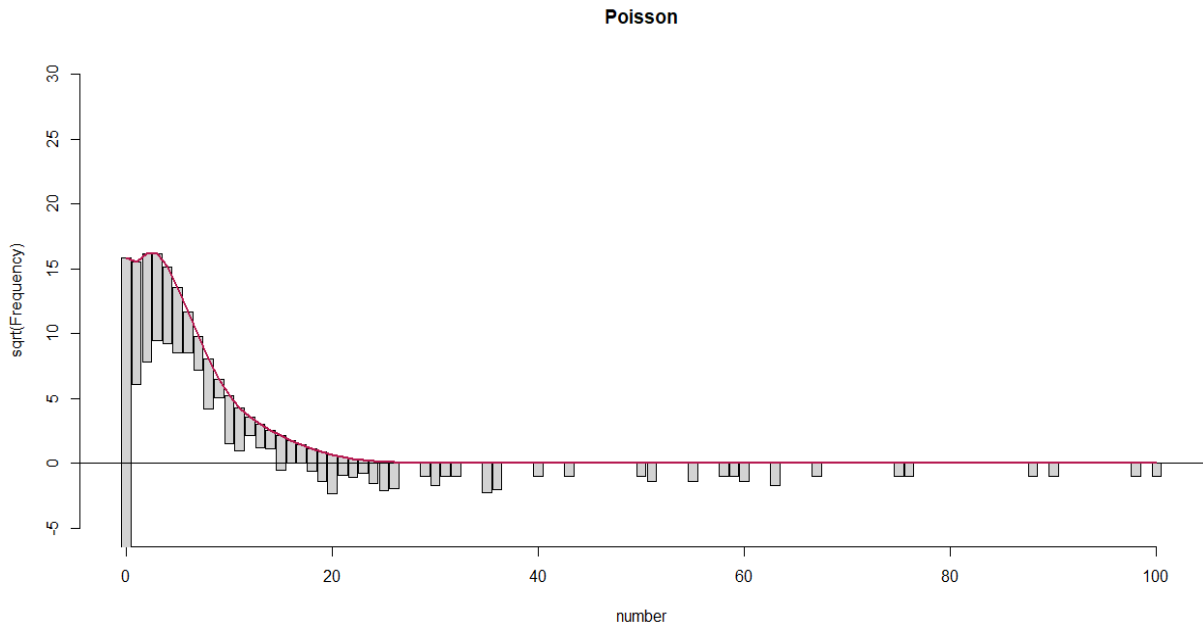
Poisson

Figure 7.19. Rootogram showing the Poisson distribution of the 'All Species Total Abundance Vs Treatment' dataset. Strong over and underpredictions of most counts indicate that overdispersion exists and excess zeros are unaccounted for.

Simulated values, red line = fitted model. p-value (two.sided) = 0.952
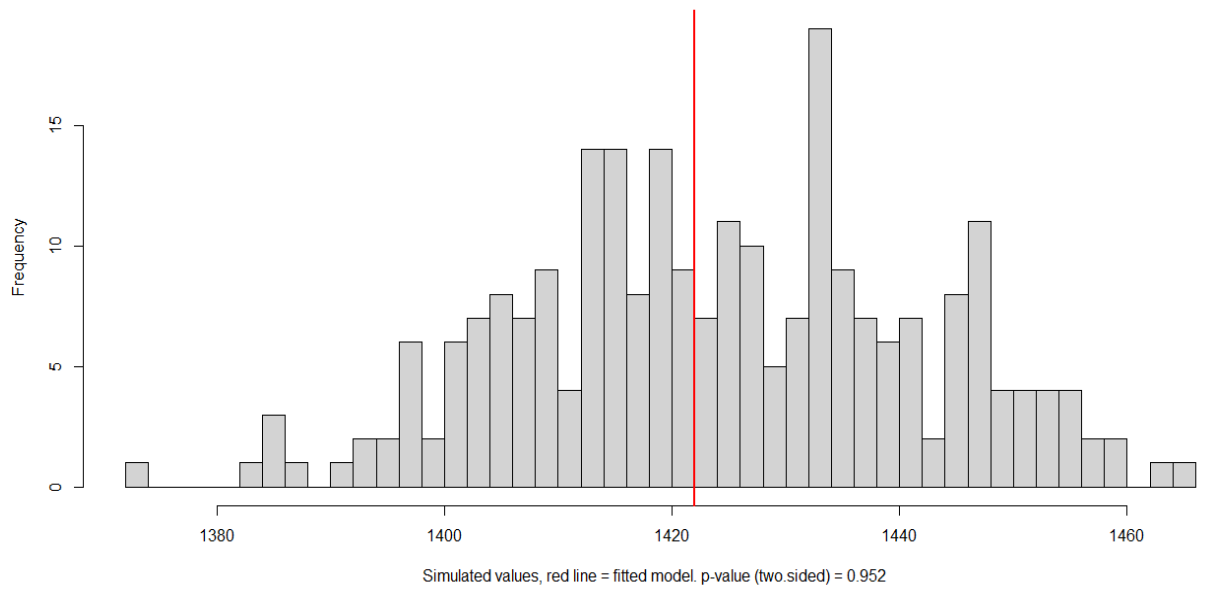
Figure 7.20. Zero-inflation plot for the 'All Species Total Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
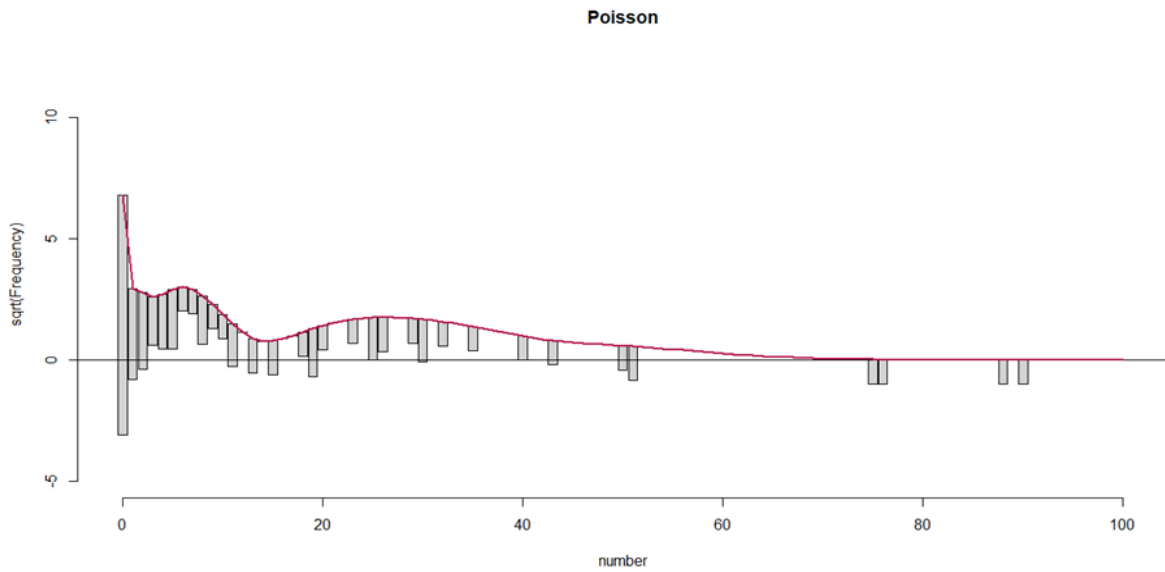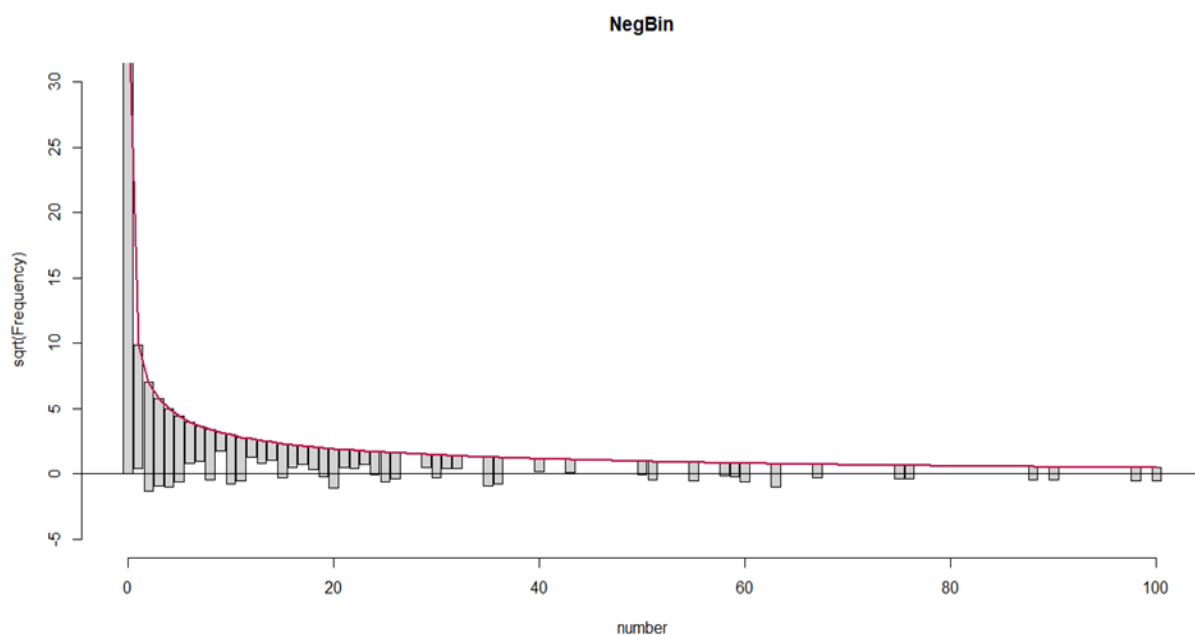
NegBin

Figure 7.21. Rootogram showing the Neg Bin distribution of the 'All Species Total Abundance Vs Treatment' dataset. Under and over predictions from expected counts are small indicating a suitably fit model, though the zero bin is large and may need to be accounted for using a zero-inflated model.

7.1.3. Species Diversity and Similarity

Table 7.1. Common names, Latin names, short codes, life histories and New Zealand (NZ) status for the 10 species present within the sampled sites of this study. * Information sourced from the New Zealand Bio-Recording Network Trust (2021).

Common Name	Latin Name*	Short Code	Life History*	NZ Status*
Inanga	*Galaxias maculatus*	inga	Amphidromous	Native
Common Bully	*Gobiomorphus cotidianus*	cmbu	Amphidromous	Endemic
Crab sp.		crab		
Goldfish	*Carassius auratus*	gldf	Resident	Introduced
Grey Mullet	*Mugil cephalus*	grml	Catadromous	Native
Mosquitofish	*Gambusia affinis*	gamb	Resident	Introduced
NZ Freshwater Shrimp	*Paratya curvirostris*	srmp	Anadromous	Endemic
NZ Longfin Eel	*Anguilla dieffenbachii*	leel	Catadromous	Endemic
Shortfin Eel	*Anguilla australis*	seel	Catadromous	Native
Yellow-eye Mullet	*Aldrichetta forsteri*	ylml	Catadromous	Native

7.1.4. The Effects of Environmental Factors on Species Abundance

7.1.4.1. Inanga Abundance Vs Barrier Status
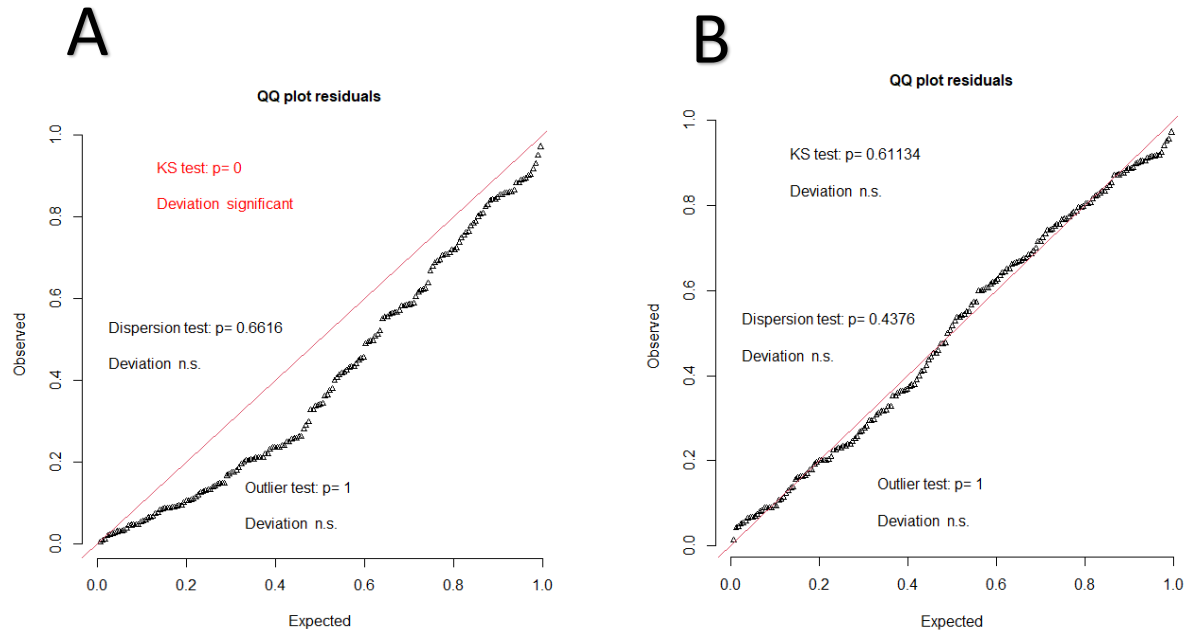
Figure 7.22. Comparison of Poisson and negative binomial distributions of 'Inanga Abundance Vs Barrier Status' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLMM for the 'Inanga Abundance Vs Barrier Status' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B**: Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'Inanga Abundance Vs Barrier Status' dataset.
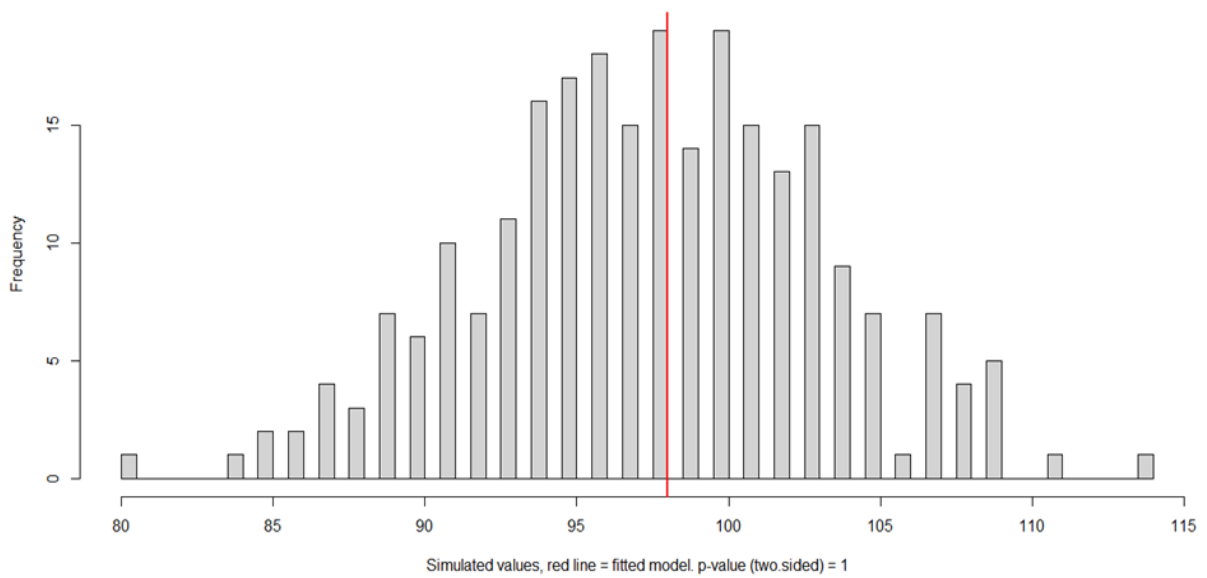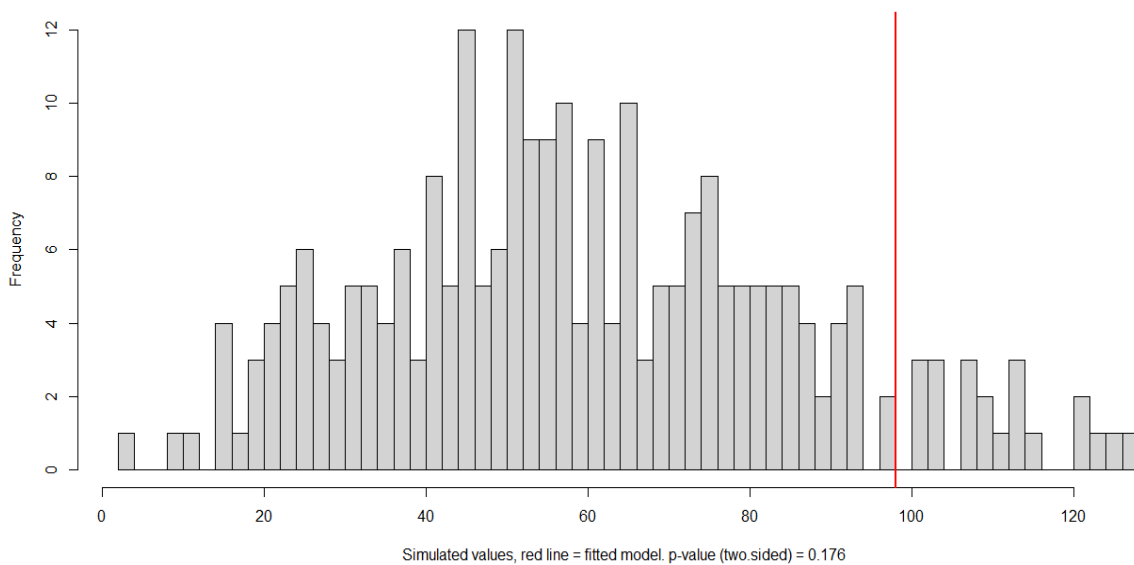

Figure 7.23. Zero-inflation plot for the 'Inanga Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
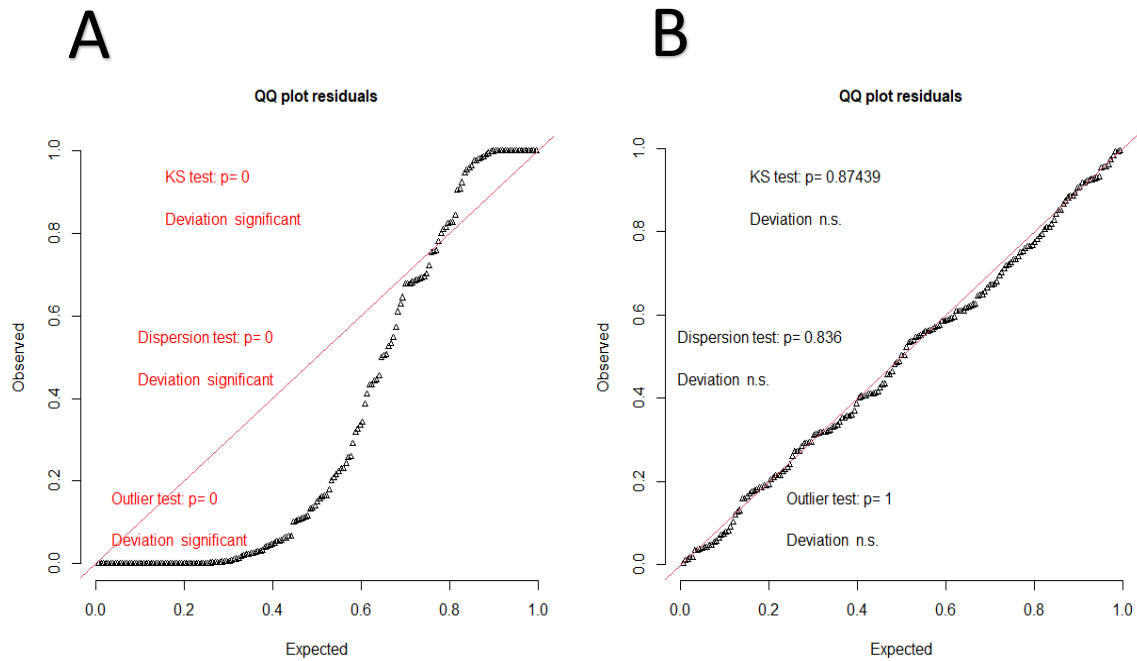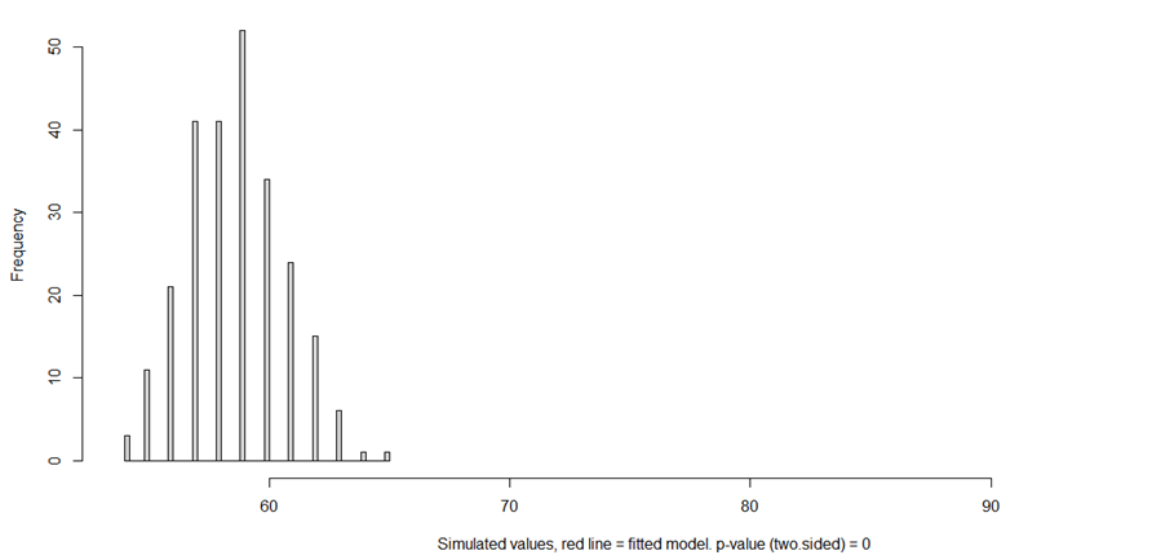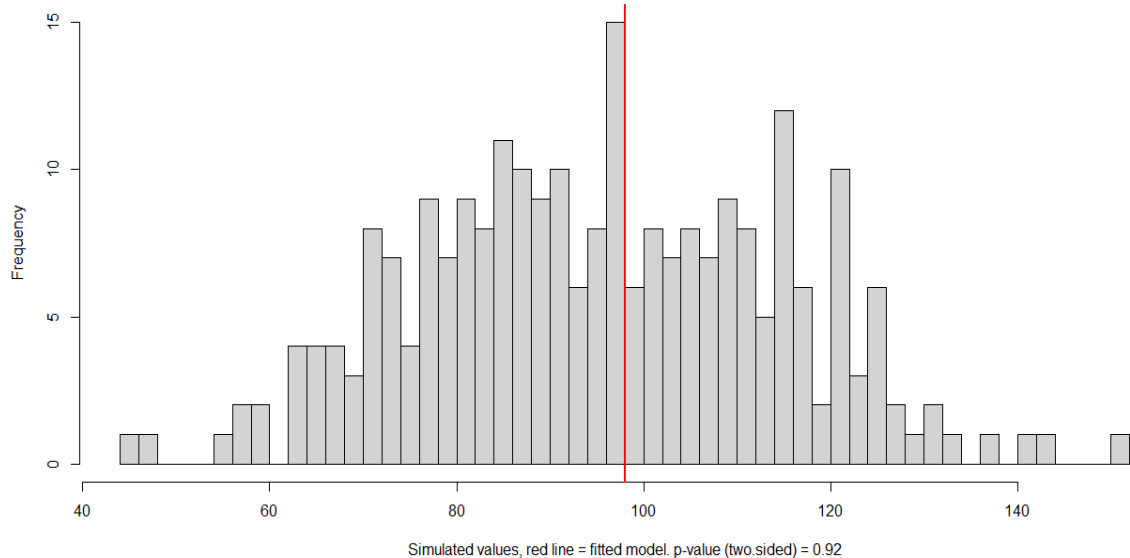
Figure 7.24. Zero-inflation plot for the 'Inanga Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
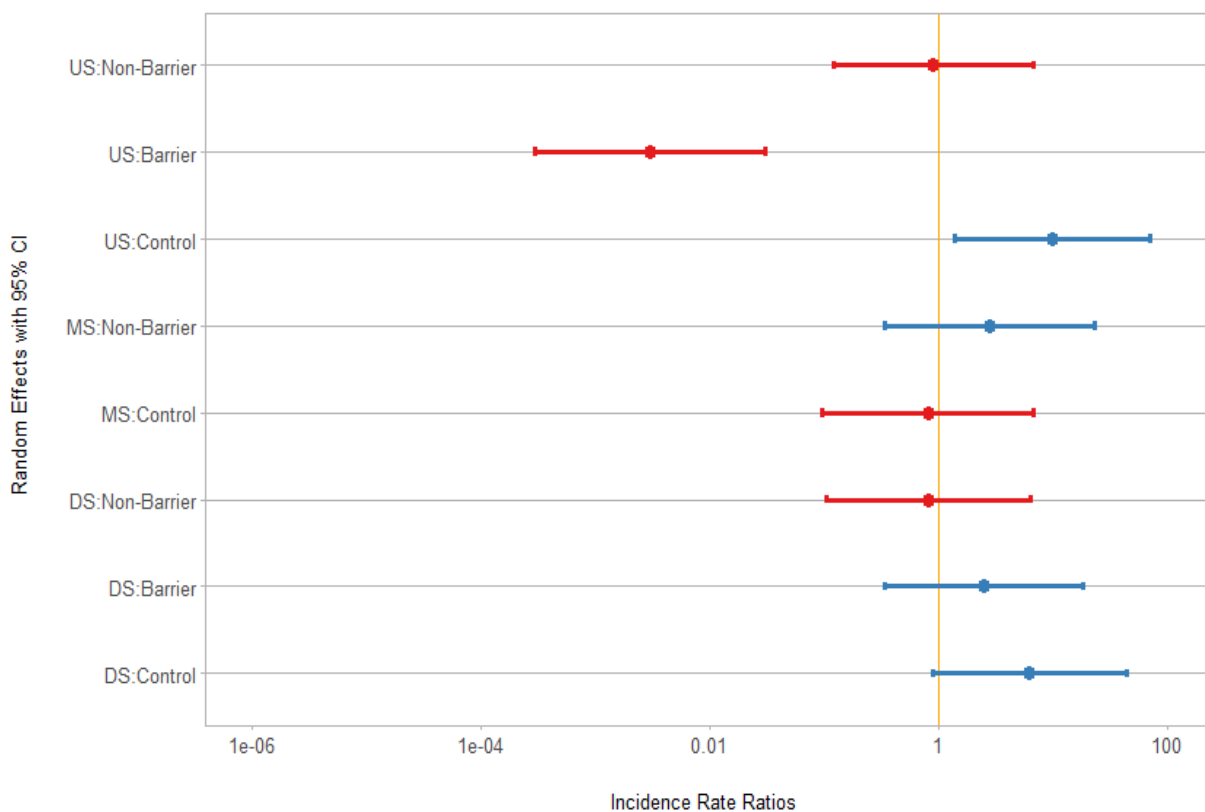
Figure 7.25. Incidence rate ratios (IRR) for the nested barrier status and position random effect (RE) variables with 95% confidence intervals (CI) for the conditional mixed effect model, using the 'Inanga Abundance Vs Barrier Status' dataset. REs in red below the orange 'baseline' indicate IRR values of <1 and a decrease in abundance, whilst REs in blue above the 'baseline' indicate IRR values of >1 and an increase in abundance.

7.1.4.2. Inanga Abundance Vs Treatment
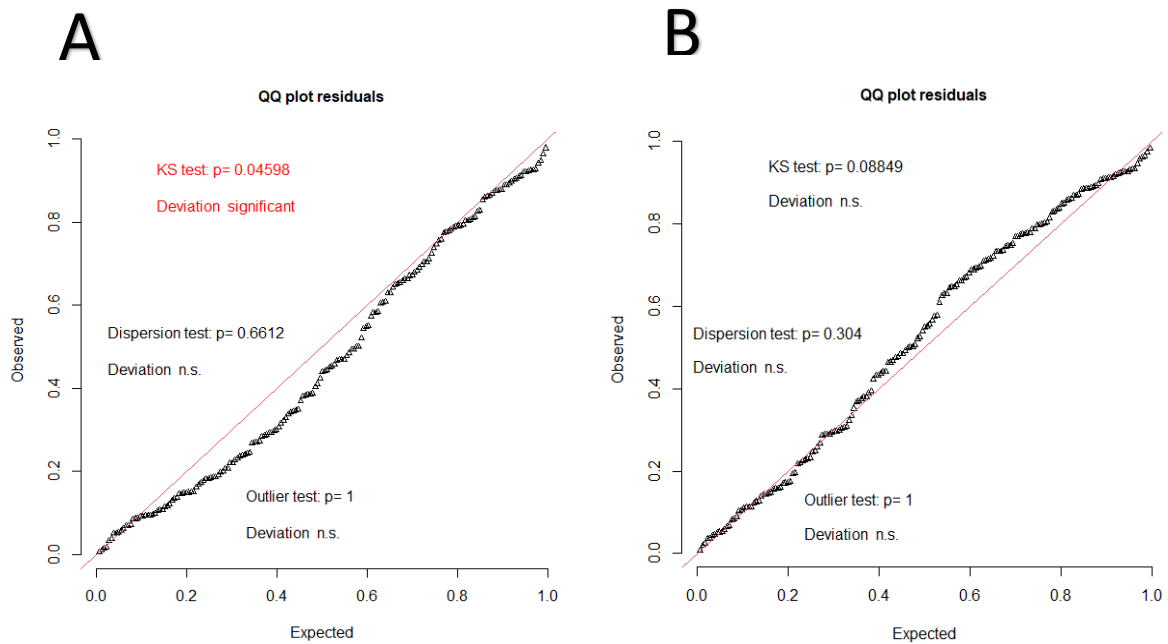
Figure 7.26. Comparison of Poisson and negative binomial distributions of 'Inanga Abundance Vs Treatment' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLMM for the 'Inanga Abundance Vs Treatment' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B**: Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'Inanga Abundance Vs Treatment' dataset.
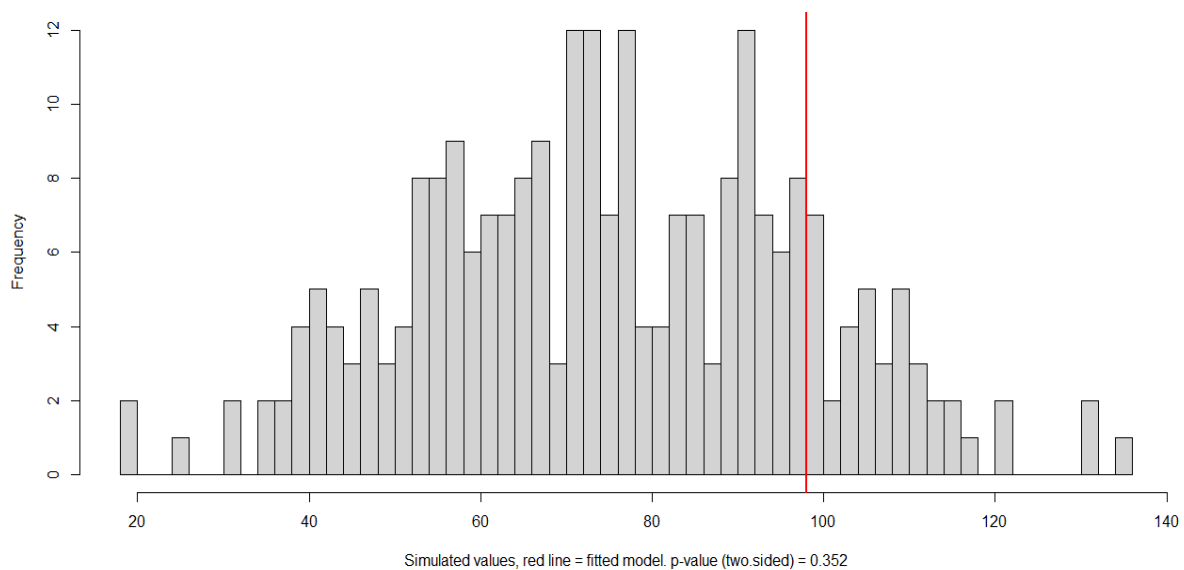
Figure 7.27. Zero-inflation plot for the 'Inanga Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
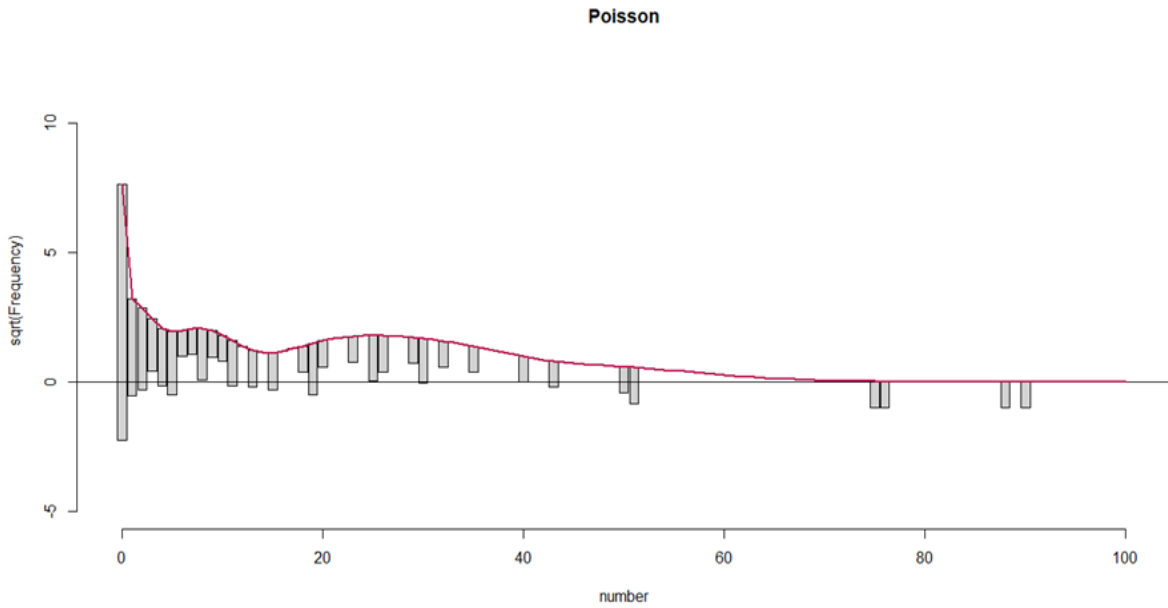
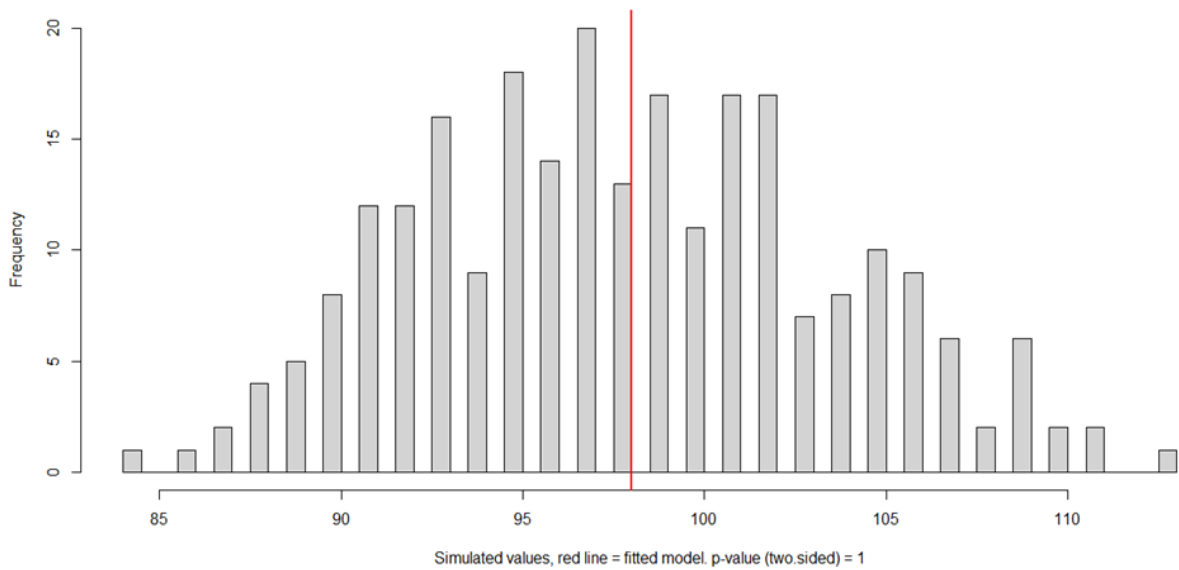Simulated values, red line = fitted model. p-value (two.sided) = 0.72
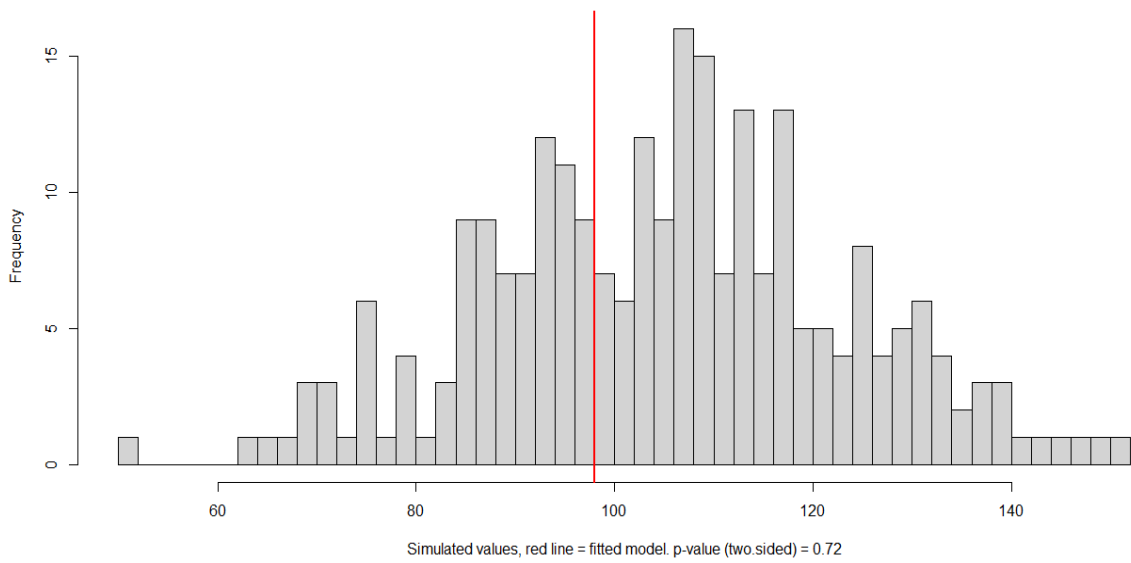
Figure 7.28. Zero-inflation plot for the 'Inanga Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
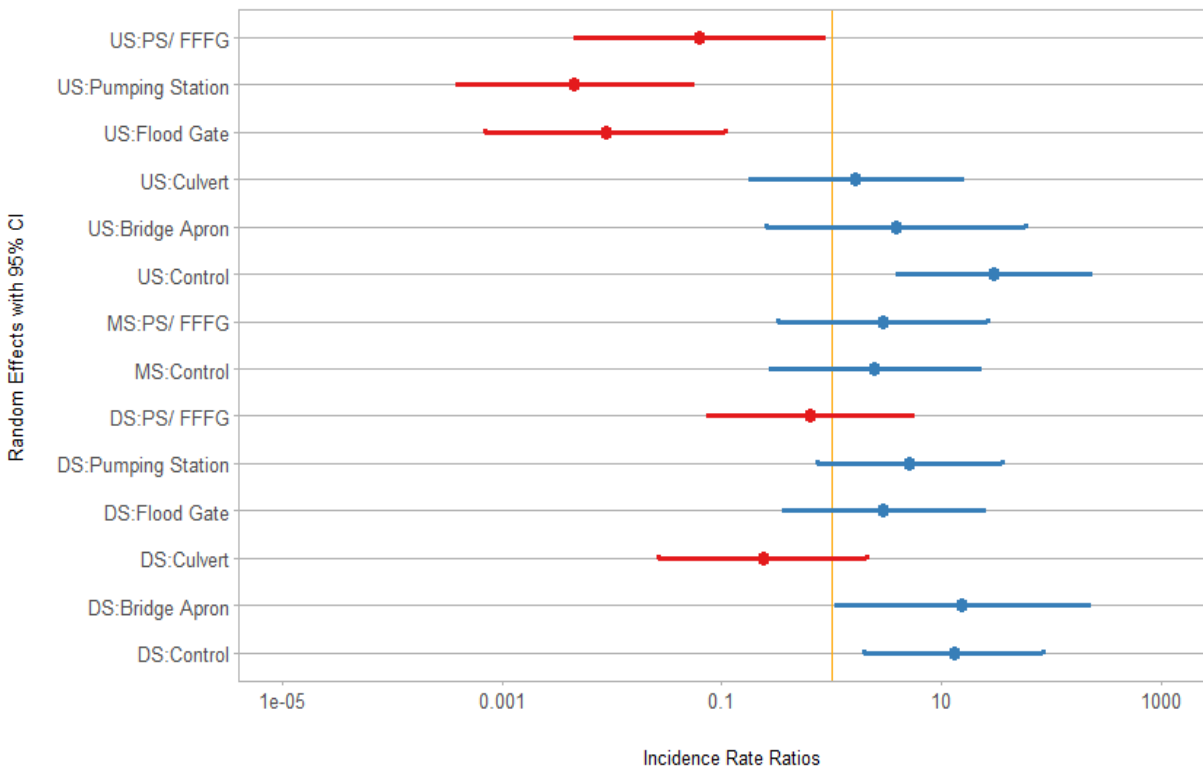

Figure 7.29. Incidence rate ratios (IRR) for the nested treatment and position random effect (RE) variables with 95% confidence intervals (CI) for the conditional mixed effect model, using the 'Inanga Abundance Vs Treatment' dataset. REs in red below the orange 'baseline' indicate IRR values of <1 and a decrease in abundance, whilst REs in blue above the 'baseline' indicate IRR values of >1 and an increase in abundance.

117

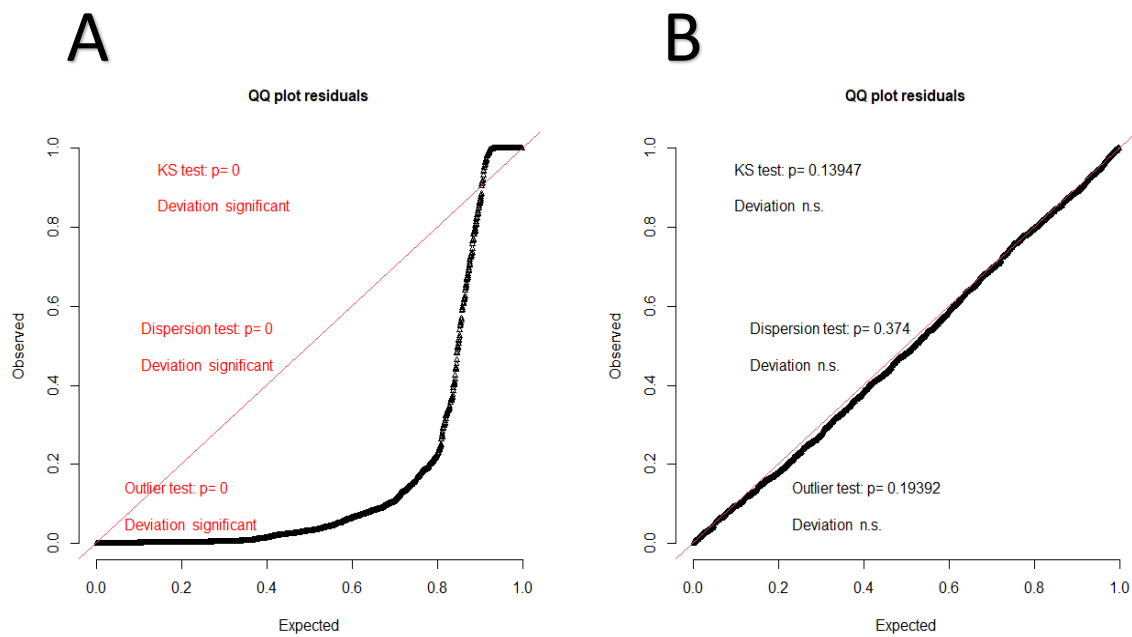### 7.1.4.3. All Species Total Abundance Vs Barrier Status
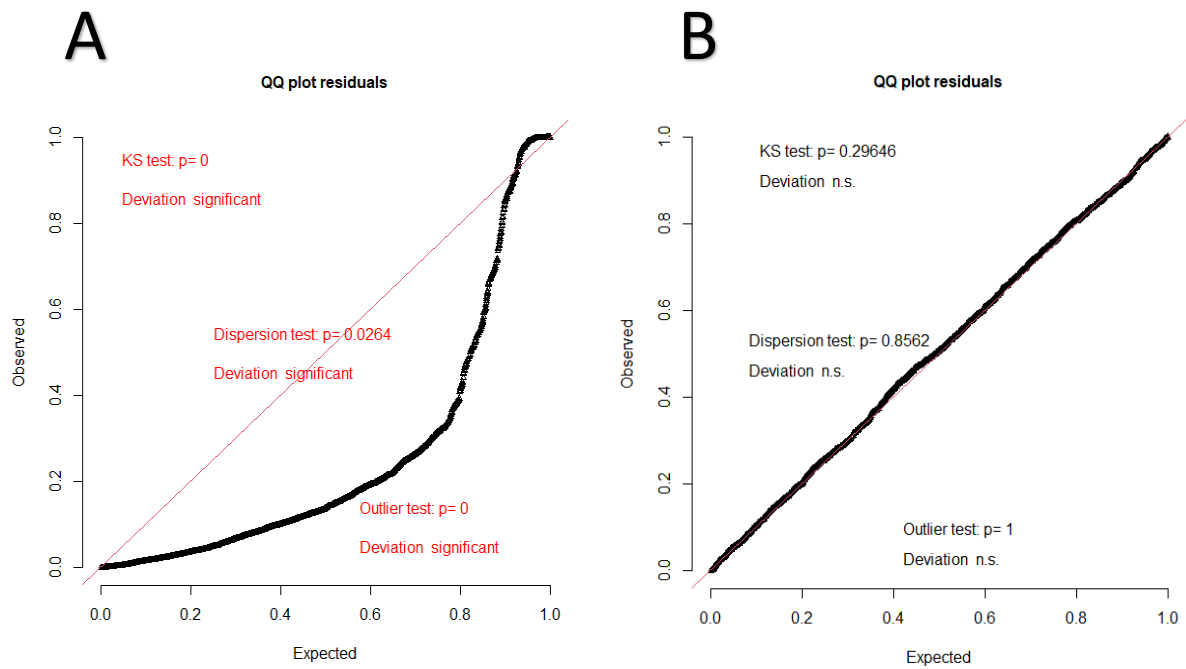
Figure 7.30. Comparison of Poisson and negative binomial distributions of 'All Species Total Abundance Vs Barrier Status' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLMM for the 'All Species Total Abundance Vs Barrier Status' dataset. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B**: Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'All Species Total Abundance Vs Barrier Status' dataset.
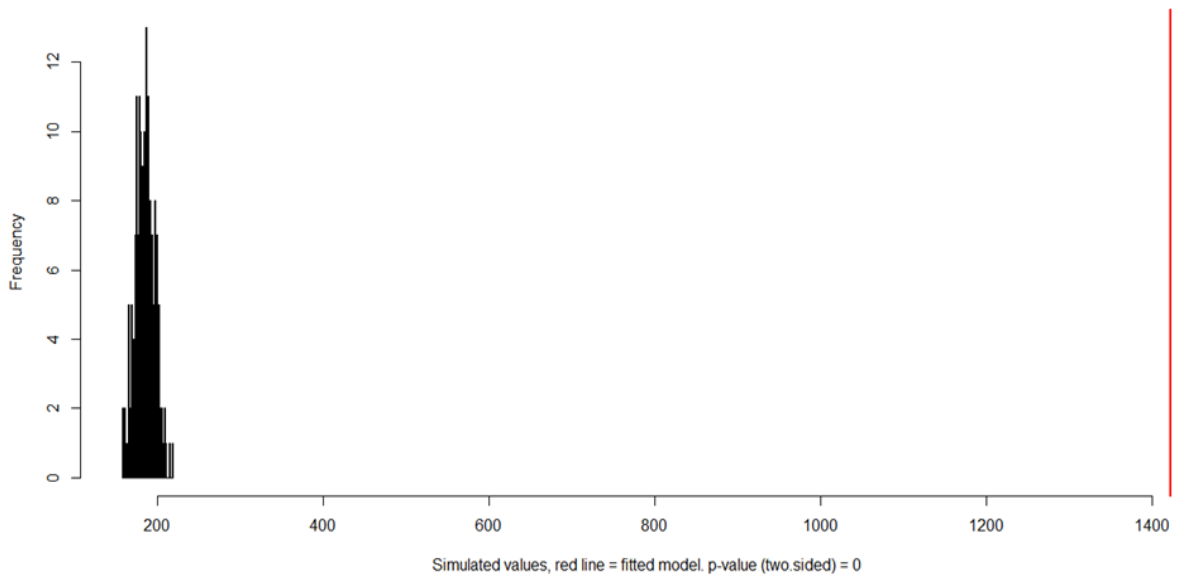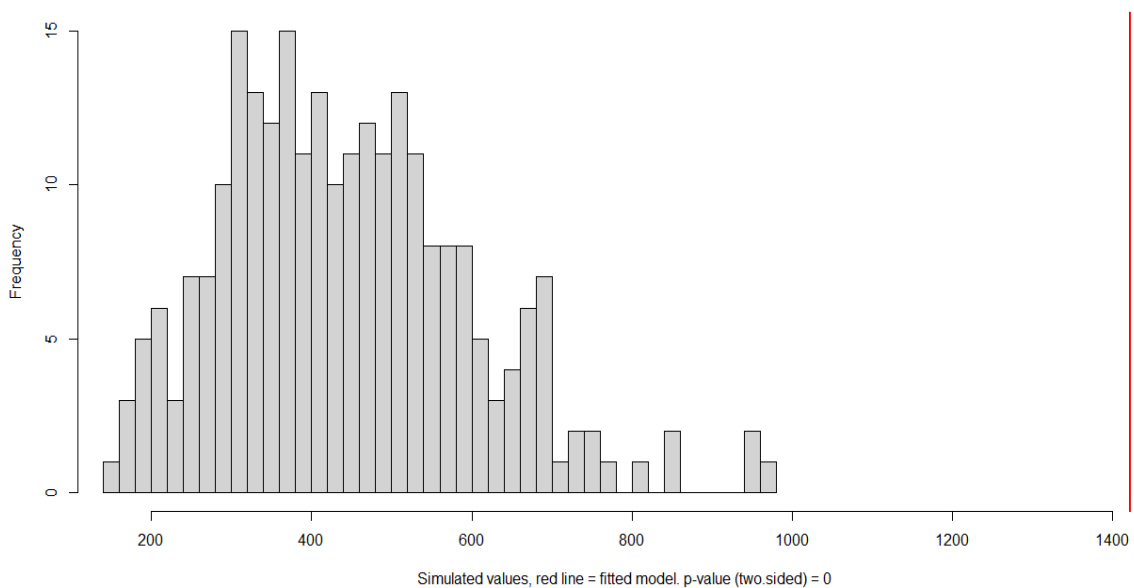

Figure 7.31. Zero-inflation plot for the 'All Species Total Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
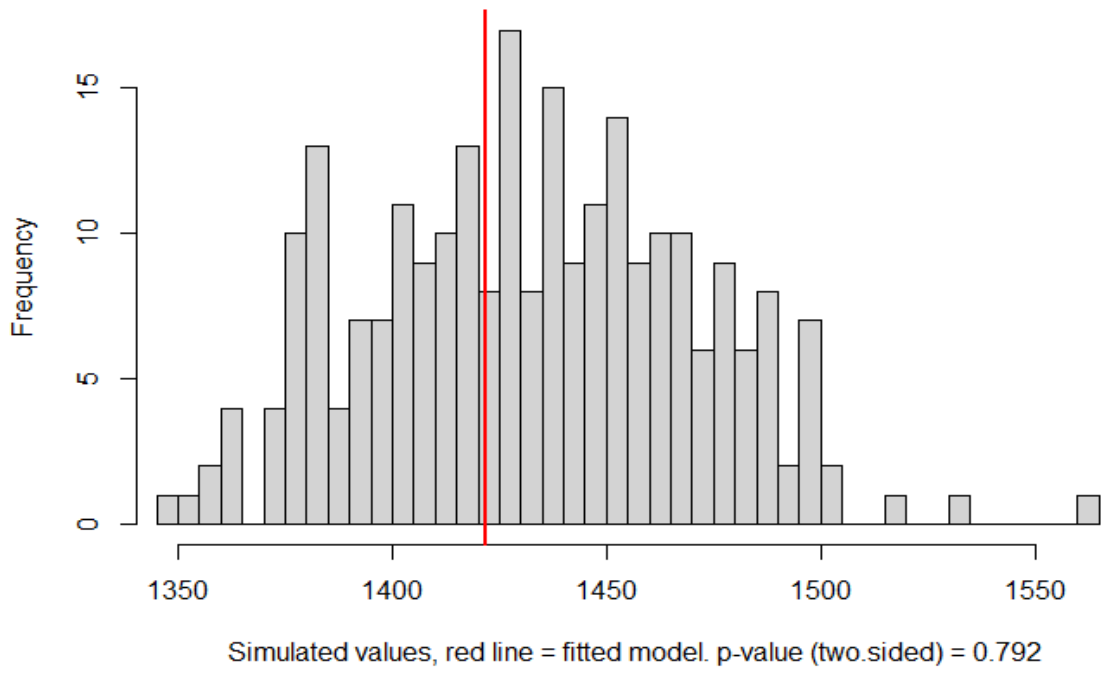
Figure 7.32. Zero-inflation plot for the 'All Species Total Abundance Vs Barrier Status' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
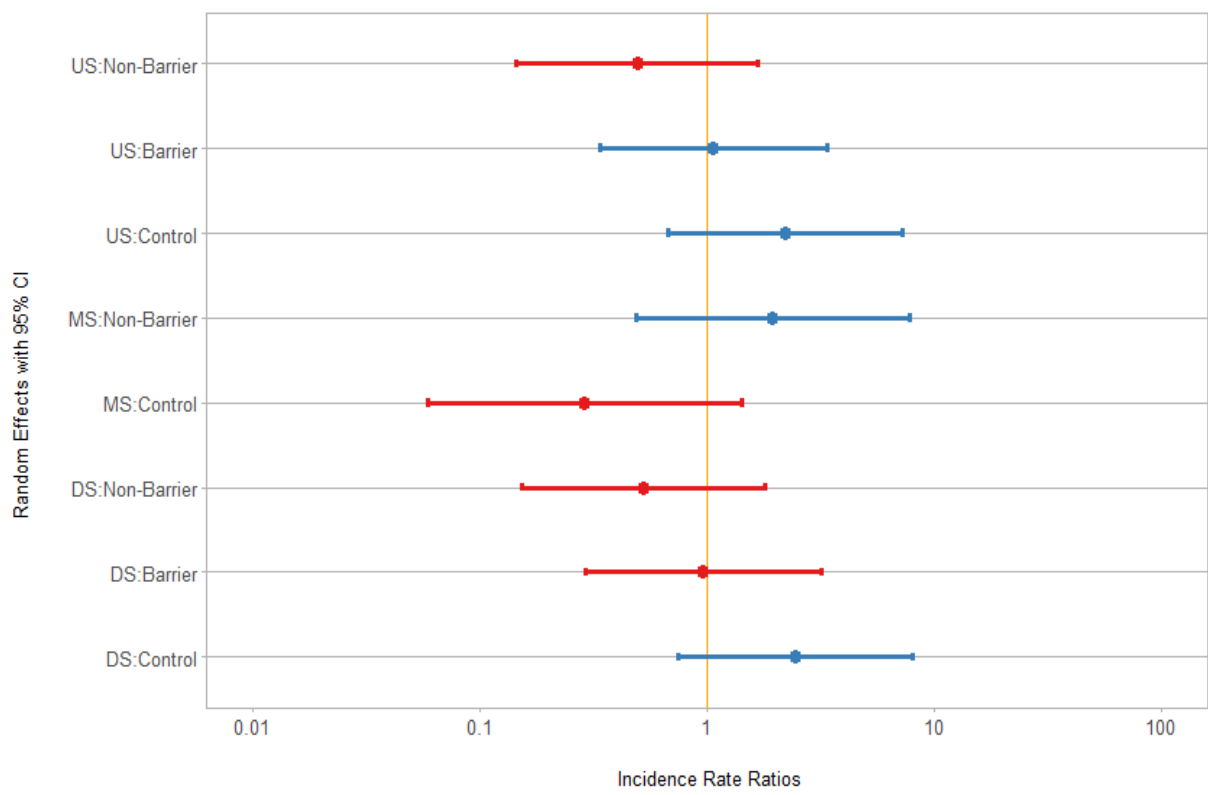

Figure 7.33. Incidence rate ratios (IRR) for the nested barrier status and position random effect (RE) variables with 95% confidence intervals (CI) for the conditional mixed effect model, using the 'All Species Total Abundance Vs Barrier Status' dataset. REs in red below the orange 'baseline' indicate IRR values of <1 and a decrease in abundance, whilst REs in blue above the 'baseline' indicate IRR values of >1 and an increase in abundance.

119

7.1.4.4. All Species Total Abundance Vs Treatment
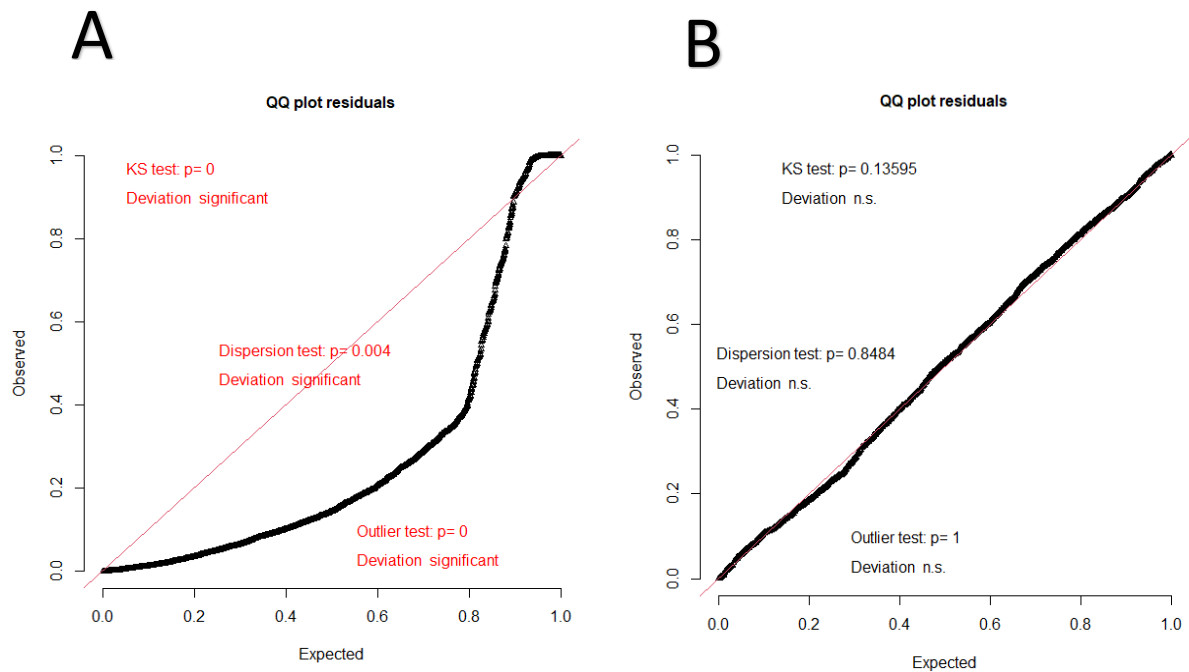
Figure 7.34*.* Comparison of Poisson and negative binomial distributions of 'All Species Total Abundance Vs Treatment' dataset.

Quantile residual QQ plots with KS, dispersion and outlier tests simulated from the GLMM for the 'All Species Total Abundance Vs Treatment'. **A:** Significant results indicate that the null, or the Poisson model is rejected in favour of an alternative model. **B**: Non-significant results indicate that the null, or the Neg Bin model is not rejected and is therefore the better fitting model for the 'All Species Total Abundance Vs Treatment' dataset.
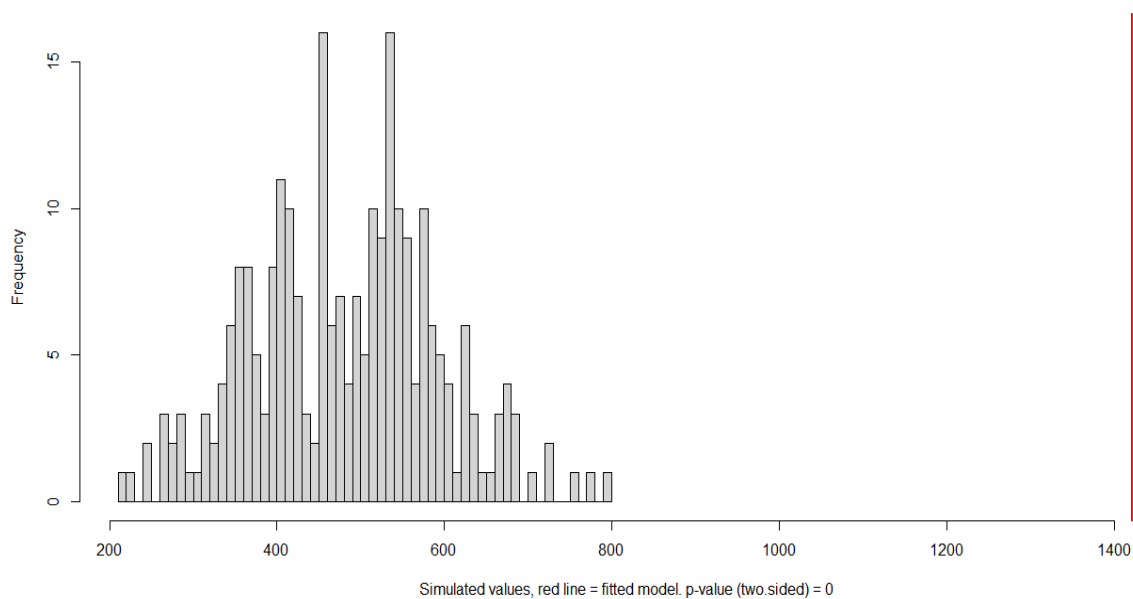
Figure 7.35*.* Zero-inflation plot for the 'All Species Total Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Poisson model, with the empirical number of zeros presented by the red line.
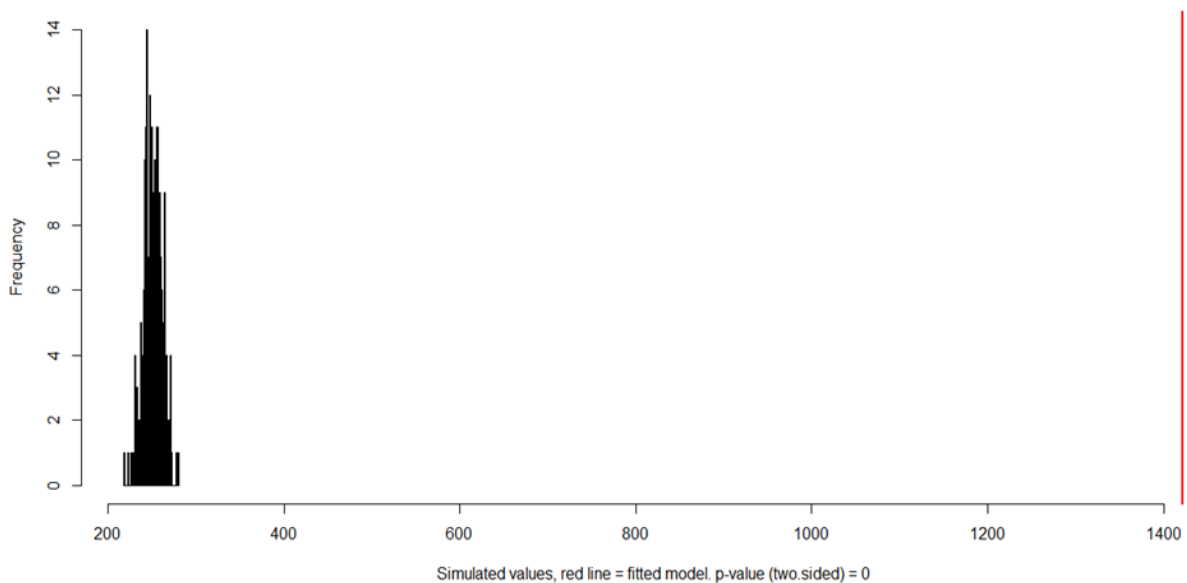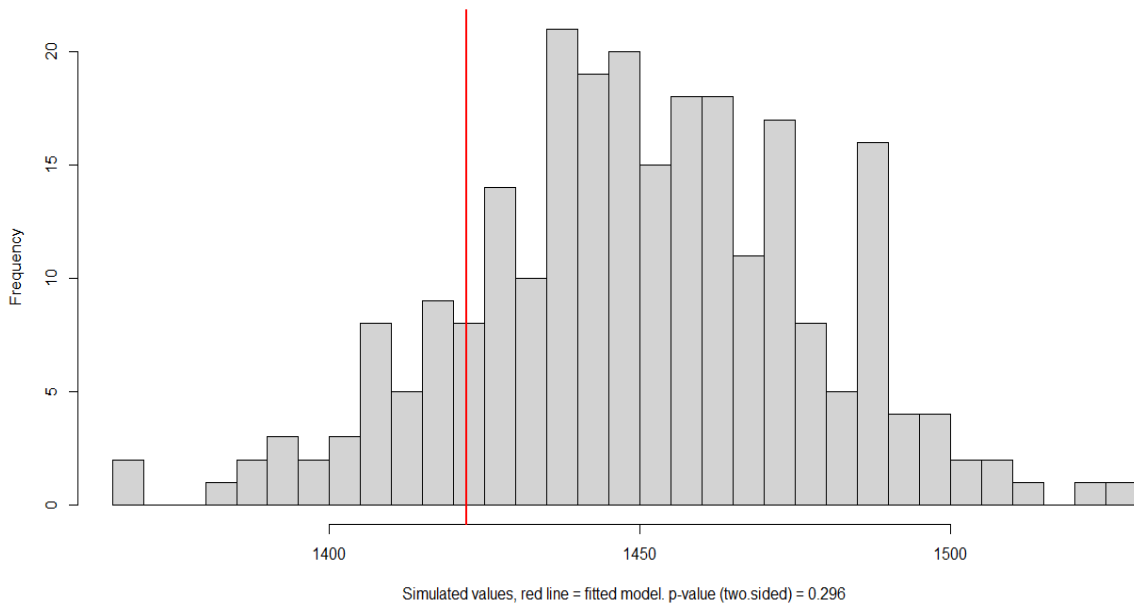
Figure 7.36. Zero-inflation plot for the 'All Species Total Abundance Vs Treatment' dataset via comparison to the expected number of simulated zeros under the null, or Neg Bin model, with the empirical number of zeros presented by the red line.
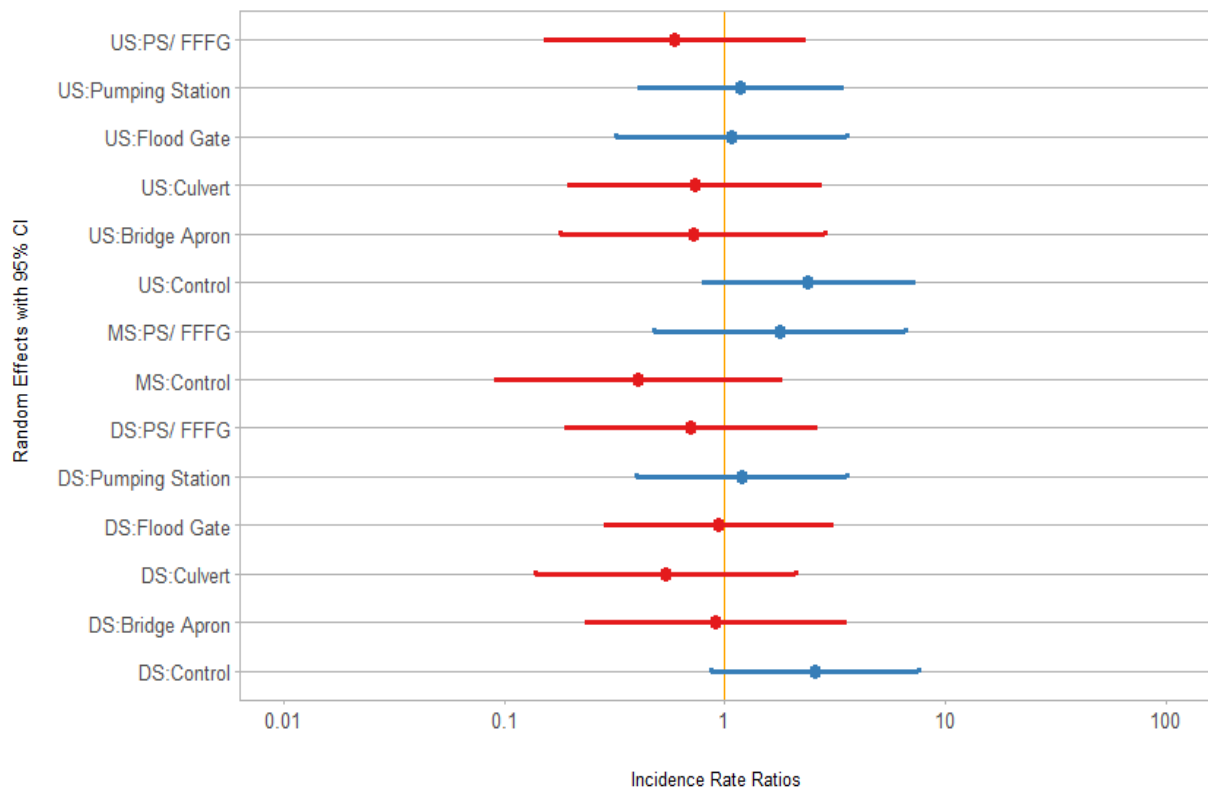
Figure 7.37. Incidence rate ratios (IRR) for the nested treatment and position random effect (RE) variables with 95% confidence intervals (CI) for the conditional mixed effect model, using the 'All Species Total Abundance Vs Treatment' dataset. REs in red below the orange 'baseline' indicate IRR values of <1 and a decrease in abundance, whilst REs in blue above the 'baseline' indicate IRR values of >1 and an increase in abundance.

7.2. Appendix B – Site Specific Management

7.2.01. Upper Taipo Stream (taup)

<u>Site Description</u>

<u>Date of Study:</u>

Thursday 12th December 2019.

<u>Weather Conditions:</u>

Dense cloud cover with breaks of sunshine in the afternoon; warm air temperature throughout the day with the occasional cool breeze.

<u>Hydrographic Tide Predictions:</u>

Non-Tidal site. Lowest tide of 0.4m at 12:35 rising to 1.6m by 18:42 (LINZ, 2020).
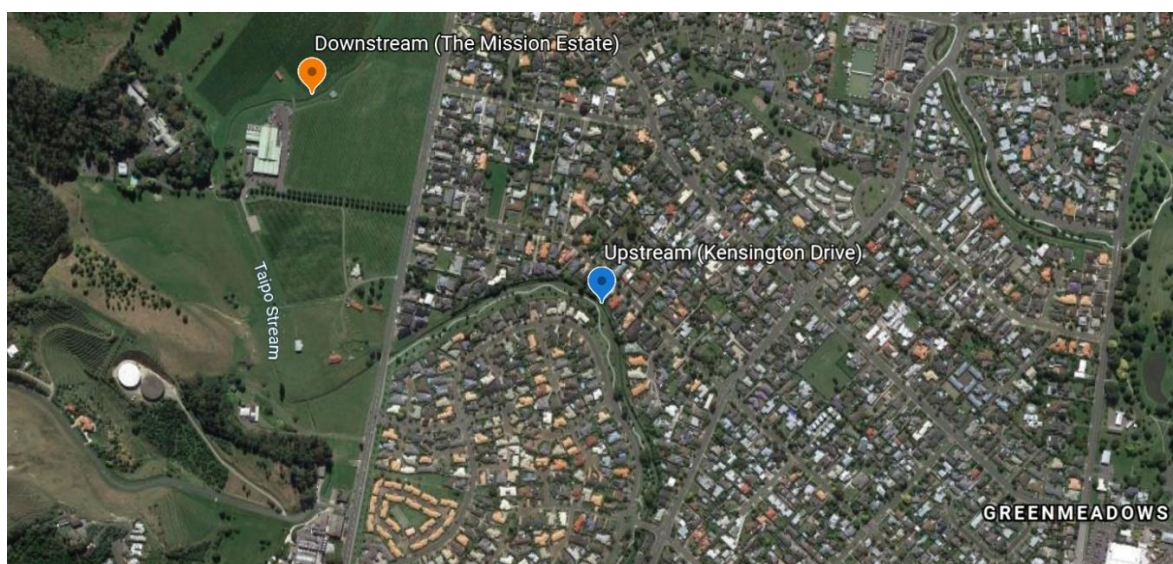
Figure 7.38. Google Earth image showing the location of the studied upstream (Kensington Drive) and downstream (The Mission Estate) sites along the upper Taipo Stream, in relation to the nearby village of Greenmeadows (Google Earth, 2021).

Branching from the Ahuriri Estuary, the Taipo Stream runs in a southerly direction through western Hawke's Bay. The downstream sub-site along the upper region of the stream lies 10.1km from the open ocean, running parallel to Church Road and continuing south through the Mission Estate's viticultural land. The stream meanders towards the upstream sub-site, which runs parallel to Kensington Drive, west of the village of Greenmeadows. The Taipo Stream continues south for an additional 2.7km into the Dolbel Nature Reserve in the sub-urban town of Taradale (Fig.7.38).

Preliminary phone calls and a subsequent in-person meeting with the Mission Estate's general manager were required to gain consent to access and set nets within the privately owned region of the stream. As the public did not have free access to the Estate's grounds, security was not an issue for the downstream sub-site. The upstream Kensington Drive sub-site was easily and openly accessible to the public, so permissions or the use of the HBRC master key were not required. Regular checking of the nets throughout the day ensured that the site's security remained uncompromised.

There are no drainage structures present along the Upper Taipo Stream, therefore this site will be used as a control.

Upstream (Kensington Drive)

GPS Coordinates (DD): -39.520885, 176.851674

Figure 7.39. The upstream habitat along the upper Taipo Stream. **A**: Heavily mown grass with sparsely scattered trees span the left bank's riparian buffer. **B**: Natural and varied vegetation spans the sub-site's right bank. **C**: Patches of shading is cast across the width of the stream. **D**: A slumping bank and active erosion beneath overhanging riparian vegetation.

There is little hydraulic heterogeneity along the upper Taipo's upstream sub-site, as the highly simplified region of the stream holds a lentic water channel only, without additional lotic components like cascades or rapids. Heavily mown grass spans the ~2m width of the left bank, with little vegetation encroaching into the stream (Fig.7.39A). Mature native trees are sparsely scattered along the left bank, casting patches of shade across the width of the stream. The right bank spans 4-5m and divides the sub-site from neighbouring private gardens. The riparian vegetation is natural and varied as native trees with an intact and diverse understory spans the stream's bank (Fig.7.39B). Mature shrubs and long exotic grasses encroach into the waterway, providing suitable resting and feeding habitat for aquatic species. Significant shading is cast across the stream along regions with the tallest trees and the densest overhanging vegetation (Fig.7.39C). Considerable bank slumping occurs below many of these areas, leading to substantial landslips and the active or recent erosion of ~35% of each bank (Fig.7.39D). Fluctuations in the water level during periods of high rainfall has also caused undercutting of the banks, leading to further erosion and bank instability.

Approximately 40% of the stream bed is covered in fine sediment, like silt or clay, although various substrate types are scattered across the sub-site. Woody debris and plant matter fallen from each

bank's riparian strips, and boulders and exposed root mats from recent landslips provide habitat for EPT invertebrate species to colonise and inhabit, as well as acting as suitable habitat for fish to shelter and spawn. Macrophytes are limited within the water channel, due to the patchy and unpredictable shading which is cast across the stream.

Downstream (The Mission Estate)

GPS Coordinates (DD): -39.51755, 176.845821

Figure 7.40. The downstream habitat along the upper Taipo Stream. **A**: Heavily mown grass spans the width of the left bank. **B**: Dense exotic shrubs and grasses span the right bank and encroach into the stream. **C**: Floating macrophytes are present in isolated patches. **D**: There is little turbulence and a slow water flow across the sub-site.

As with the upstream sub-site, the downstream sub-site along the upper Taipo Stream has little hydraulic heterogeneity as there are no lotic cascades, waterfalls or riffles present. Spanning 2-3m in width, the left bank has been heavily mown down to the water's edge, providing little or no shading across the stream, and limited suitable resting or spawning habitat for aquatic species (Fig.7.40A). Fluctuating water levels during high rainfall events have led to ~35% of the exposed left bank showing signs of active erosion, though the resulting undercut banks may provide small-bodied fish some shelter from larger-bodied predators. The right bank spans ~5m in width and holds a variety of dense exotic shrubs, sedges and long grasses (Fig.7.40B). Although some upper reaches of the right bank have been heavily mown, a thick strip of riparian vegetation remains along the length of the sub-site by the water's edge, providing ample spawning grounds for migrating and resident fish. The encroaching vegetation casts regular shading across ~15% of the waterway, whilst also acting as the bank's defence against erosion caused by fluctuating water levels.

The number of substrate types within the stream are limited; boulders and cobbles are sparsely scattered amongst silt and clay, which covers only ~40% of the stream bed. A small volume of fallen woody debris and plant matter has accumulated around the right bank, providing some suitable habitat for EPT invertebrate colonisation and additional shelter for fish species. Isolated patches of floating macrophytes span the length of the sub-site (Fig.7.40C), slowing the flow of water and reducing the amount of turbulence (Fig.7.40D).

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 728.20) yields a Pearson-chisq dispersion value of 68.77 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 120.81) yields a Pearson-chisq value of 1.00 (p = 0.44) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.67 (p = 0.25) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 4249.65) yields a Pearson-chisq dispersion value of 112.17 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 299.34) yields a Pearson-chisq statistic of 0.97 (p = 0.57) = overdispersion does not exist. An additional Vuong AIC z statistic of 3106.86 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 124) is not higher than the inanga abundance in the downstream (DS) sub-site (N = 657) along the upper Taipo stream, despite a statistically significant result (p = 0.03) (Fig.7.41). An IRR value of 0.19 for the US sub-site (95% CI: 0.04 – 0.84) is not higher than an IRR value of 109.50 for the DS sub-site (95% CI: 38.20 – 313.84), showing that inanga abundances are higher downstream, despite no structure being present between the two sub-sites.

The total abundance of all species in the US sub-site (N = 135) is not significantly higher than the total abundance of all species in the DS sub-site (N = 678) along the upper Taipo stream (p = 0.06) (Fig.7.41). An IRR value of 0.20 for the US sub-site (95% CI: 0.04 – 1.05) is not higher than an IRR value of 11.30 for the DS sub-site (95% CI: 3.50 – 36.46), showing that a higher total abundance of all species inhabit the downstream reaches of the stream, despite no barrier structure being present.
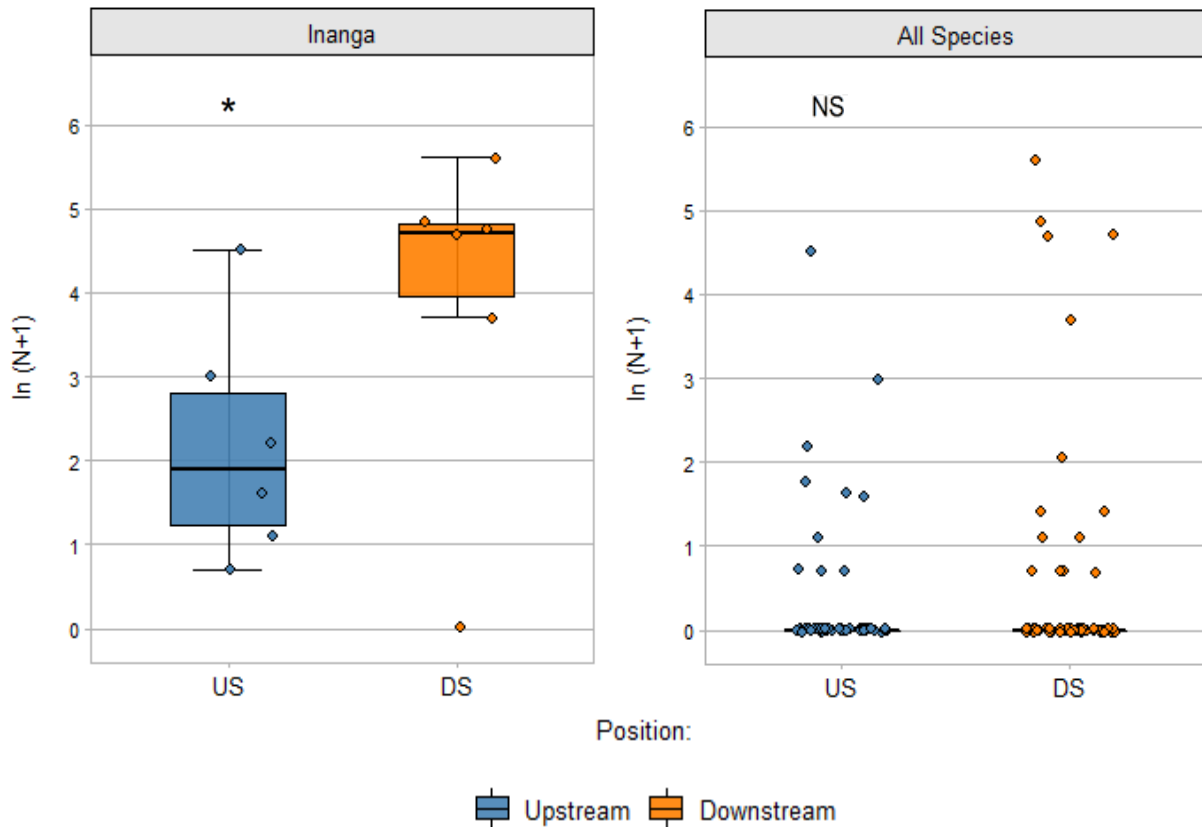
Figure 7.41. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the upper Taipo Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and 'All Species').

Species Diversity and Similarity

H_0: *Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.*

H_1: *Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.*

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.31) along the upper Taipo stream is not significantly different to the H' value of the DS sub-site (H' = 0.16) as an US ENS value of 1.36 (H'_{max} = 1.10, p = 5.18E-02) and a DS ENS value of 1.18 (H'_{max} = 1.39) shows that a similar number of species are present within both upstream and downstream reaches of the stream and that upstream fish passage is unrestricted.

Pielou's evenness index (J') within the US sub-site (J' = 0.28) is higher than the J' of the DS sub-site (J' = 0.12), although the US species richness (S = 3) was marginally lower than the DS species richness (S = 4). A Sørensen's similarity index of 0.57 indicates that 57% of the species present across the stream are found within both sub-sites. The migratory inanga and the invasive mosquitofish are the two species present within both upstream and downstream reaches, whereas the migratory shortfin eel and the invasive goldfish are present within the DS sub-site only.

126

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature and specific conductivity measures are similar within both sub-sites, although the dissolved oxygen saturation was considerably lower downstream. As there is no barrier structure present along the stream, the habitat conditions of the downstream sub-site alone have led to the low observed levels of DO. A higher macrophyte density and deeper pools along the downstream sub-site may have led to the decline in DO and provided the optimal conditions for a larger abundance of inanga to thrive, compared with the upstream sub-site.

7.2.02. Lower Taipo Stream (talm)

Site Description

Date of Study:

Friday 13th December 2019.

Weather Conditions:

Dense cloud in the morning changing to clear skies and high sunshine levels by the afternoon; increasing air temperatures into the afternoon.

Hydrographic Tide Predictions:

Tidal site. Lowest tide of 0.3m at 13:22 rising to 1.6m by 19:30 (LINZ, 2020).
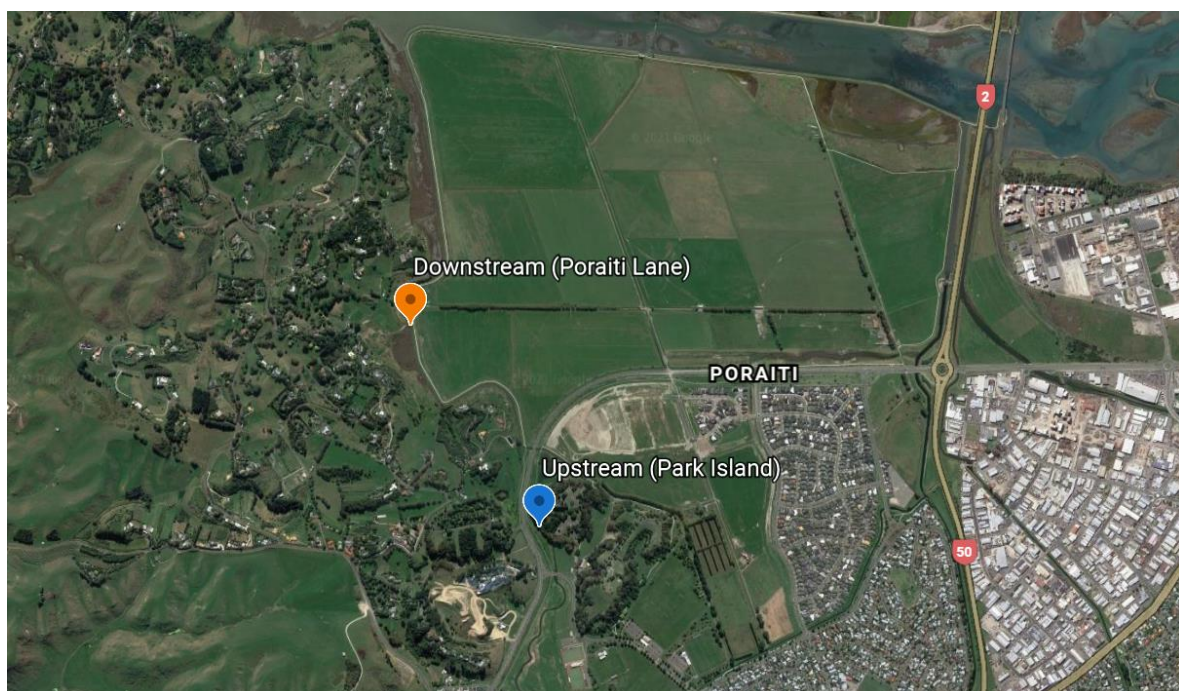
Figure 7.42. Google Earth image showing the location of the studied upstream (Park Island) and downstream (Poraiti Lane) sites along the lower Taipo Stream, in relation to the nearby village of Poraiti and the Ahuriri Estuary (Google Earth, 2021).

Branching from the Ahuriri Estuary, the lower region of the Taipo Stream runs in a southerly direction through agricultural land, west of the sub-urban village of Poraiti in western Hawke's Bay. The downstream sub-site lies 1.4km from the Ahuriri Estuary and 5.9km from the open ocean and runs perpendicular to a popular public cycle path along Poraiti Lane (Fig.7.42). The upstream sub-site continues south, parallel to Prebensen Drive and the public Park Island Nature Reserve. Approximately 2.5km lies between the lower Taipo Stream's upstream sub-site and the upper region's downstream sub-site within the Mission Estate. The Taipo Stream continues for an additional 4km above the Mission Estate's sub-site, totalling 6.5km of available upstream habitat above the Park Island sub-site.

Both sub-sites ran through public areas, therefore access did not require a permit or landowner permissions. The HBRC master key was required to gain vehicle access to the downstream sub-site. Caution was essential whilst driving along the public cycle path when pedestrians or cyclists were

within close proximity to the vehicle. Regular checking of the nets throughout the day along both sub-sites ensured that security remained uncompromised.

Drainage structure present:

There are no drainage structures present along the Lower Taipo Stream, therefore this site will be used as a control.

Upstream (Park Island)

GPS Coordinates (DD): -39.503154, 176.853778

Figure 7.43. The upstream habitat along the lower Taipo Stream. **A**: A diverse riparian strip lies below heavily mown grass along the left bank. **B**: Mature overhanging trees are numerous along the right bank. **C**: Patches of aquatic plants and long grasses provide additional fish spawning habitat. **D**: 90% of the water channel is choked with high volumes of exotic macrophytes.

Despite the upstream sub-site along the lower Taipo Stream being tidal and exposed to fluctuating water levels, there is limited hydraulic heterogeneity as the lentic water channel has little turbulence and no lotic components like riffles or cascades. Heavily mown grass spans much of the ~3m wide left bank, though a natural and diverse riparian strip, 0.5-1m in width, lines the water's edge and encroaches into the stream (Fig.7.43A). The partially submerged, dense strip contains a mixture of aquatic plants, sedges and young grasses, providing an abundance of suitable shelter and spawning habitat for migrating fish, as well as acting as the bank's defence against erosion caused by changing water levels. Spanning 2-3m in width, the right bank's riparian vegetation differs greatly from the left, as mature native trees overhang across the stream, perching on a steep muddy bank (Fig.7.43B). Bare, impervious ground lies beneath the tree canopy, which has led to significant slumping and recent erosion along ~50% of the bank. Subsequently, woody debris and small boulders and cobbles

have fallen into the water channel, exposing root mats and undercutting the bank in some areas. The fallen substrates, as well as several isolated patches of aquatic plants and long grasses offer some additional fish spawning habitat along the right bank (Fig.7.43C).

Although the large overhanging vegetation casts significant shade across ~70% of the stream, macrophyte growth remains high. As the sub-site is geographically positioned in an east to westerly direction, the stream remains in direct view of sunlight for much of the day, providing effective conditions for excessive macrophyte growth. Approximately 90% of the waterway is choked with high volumes of hornwort *(Ceratophyllum demersum)*, water buttercup *(Ranunculus* sp*.),* fool's watercress *(Apium nodiflorum)* and curled pondweed *(Potamogeton crispus),* providing copious amounts of favourable resting, feeding and shelter habitat for aquatic species (Fig.7.43D).

Downstream (Poraiti Lane)

GPS Coordinates (DD): -39.49531, 176.847009

Figure 7.44. The downstream habitat along the lower Taipo Stream. **A:** Native sedges and tussocks form a wetland habitat along the left bank during a high tide. **B**: Heavily mown grass spans the width of the right bank. **C**: Isolated patches of tussocks and brambles are scattered along the stream's edge. **D**: Silt and clay covers the stream bed and provides habitat for numerous aquatic species.

The downstream sub-site along the lower Taipo Stream has little hydraulic heterogeneity during a low tide, though small lentic pools form within each bank's riparian vegetation during a higher tide. Native sedges and tussocks span the ~4m width of the left bank, encroaching into the stream and forming a wetland environment as the water levels increase, providing an abundance of suitable shelter and spawning habitat for migrating and resident fish (Fig.7.44A). The dense layer of vegetation acts as the left bank's defence against erosion from the daily tidal fluctuations. Heavily mown grass spans the ~2m width of the right bank which offers little protection against scouring at

the water line and has led to the erosion of ~35% of the bank (Fig.7.44B). Submerged patches of mature tussocks and brambles are scattered along the stream's edge, offering some additional fish habitat, though during a low tide, the stream recedes and the vegetation becomes exposed (Fig.7.44C).

At least 75% of the stream bed is covered by a compacted layer of fine sediment, comprising of silt and clay, offering shelter for aquatic species like the endemic tunnelling mud crab *(Austrohelice crassa)* (Fig.7.44D). Boulders and cobbles are sparsely scattered across ~25% of the stream bed and provide suitable habitat for EPT invertebrate colonisation around the water's edge when the tide recedes. A small volume of woody debris and plant matter has fallen into the water channel, becoming entangled in both banks' riparian vegetation and offering additional fish shelter habitat. Although shading is cast across ~25% of the stream by each bank, macrophyte growth remains significantly lower than that found within the upstream sub-site. Floating and submerged macrophytes offer fish shelter across approximately 10% of the stream, whilst slowing the flow of water and reducing the amount of turbulence as the tide fluctuates.

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 169.09) yields a Pearson-chisq dispersion value of 16.73 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 76.70) yields a Pearson-chisq value of 1.19 (p = 0.29) = overdispersion is minimal. An additional Vuong AIC z statistic of -0.62 (p = 0.27) shows that the **zero-inflated negative binomial GLM** (AIC = 73.99) is favoured over a negative binomial GLM as zero-inflation does occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 1216.02) yields a Pearson-chisq dispersion value of 16.09 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 422.12) yields a Pearson-chisq statistic of 0.87 (p = 0.85) = overdispersion does not exist. An additional Vuong AIC z statistic of -0.35 (p = 0.36) shows that the **zero-inflated negative binomial GLM** (AIC = 420.50) is favoured over a negative binomial GLM as zero-inflation does occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 28) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 71) along the lower Taipo stream (p = 0.87) (Fig.7.45). An IRR value of 1.13 for the US sub-site (95% CI: 0.26 – 5.01) is not higher than an IRR value of 11.83 for the DS sub-site (95% CI: 5.63 – 24.86), showing that the abundance of inanga is higher downstream, despite no structure being present between the two sub-sites. The estimated

odds ratio (OR) of obtaining an excess zero in the US abundance data was considerably larger than the OR of the DS data (OR = 3.07E-09).

The total abundance of all species in the US sub-site (N = 148) is not significantly higher than the total abundance of all species in the DS sub-site (N = 201) along the lower Taipo stream (p = 0.52) (Fig.7.45). An IRR value of 1.38 for the US sub-site (95% CI: 0.52 – 3.69) is not higher than an IRR value of 3.96 for the DS sub-site (95% CI: 1.44 – 10.92), showing that a higher total abundance of all species inhabit the downstream reaches of the stream, although the difference in abundances is not considerable (Fig.7.45). The OR of obtaining an excess zero in the US abundance data is 6.69 times the OR of the DS data (OR = 0.18, p = 0.56).

Figure 7.45. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the lower Taipo Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (zero-inflated negative binomial GLM for 'Inanga' and 'All Species'). Significance level without brackets indicate count model p-values, significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 1.07) along the lower Taipo stream is significantly different to the H' value of the DS sub-site (H' = 1.55) as an US ENS value of 2.91 (H'$_{max}$

= 1.61, p = 4.40E-08) and a DS ENS value of 4.71 (H'_{max} = 1.95) shows that a greater number of species inhabit the downstream reaches of the stream, which is to be expected as the sub-site lies along the freshwater-saltwater boundary zone, an ecotone which is typically associated with high fish biodiversity.

Pielou's evenness index (J') and species richness (S) are marginally lower in the US sub-site (J' = 0.66, S = 5) compared with the DS (J' = 0.80, S = 7), whilst a Sørensen's similarity index of 0.67 indicates that 67% of species present across the stream are found within both sub-sites. The migratory inanga, shortfin eel, crab and the invasive mosquitofish are the four species present within both of the US and the DS sub-sites, whereas other migratory species like grey and yellow-eye mullet are present within the DS sub-site only.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, the mean dissolved oxygen saturation and the mean specific conductivity measures are higher within the downstream sub-site. Little shading and high exposure to direct sunlight has led to increased downstream water temperatures, though a medium macrophyte density has led to a high saturation of DO, conditions which have provided the optimal conditions for a high abundance and diversity of native, migrating fish to inhabit. Although no drainage structure is present along the stream, the excessive growth of submerged macrophytes within the upstream sub-site may act as a natural barrier to some upstream migrating fish and may provide unfavourable conditions for inhabitation.

7.2.03. Grange Creek (grcr)

Site Description

Date of Study:

Upstream: Wednesday 8th January 2020.

Downstream: Tuesday 7th January 2020.

Note: As the Grange Creek site geographically shares its downstream sub-site with the Memorial Park Drain site, the Memorial Park Drain downstream sub-site will be used as the Grange Creek downstream sub-site in this study.

Weather Conditions:

Both studied days had high sunshine levels and warm air temperatures with occasional patchy cloud and a cool breeze.

Hydrographic Tide Predictions:

Upstream: Tidal site. Lowest tide of 0.4m at 10:22 rising to 1.5m by 16:28 (LINZ, 2020).

Downstream: Tidal site. Lowest tide of 0.5m at 09:33 rising to 1.5m by 15:40 (LINZ, 2020).

Figure 7.46. Google Earth image showing the location of the studied upstream and downstream Grange Creek sub-sites in relation to the Tukituki River and the open ocean (Google Earth, 2021).

Branching south from the Tukituki River, Grange Creek runs in a south westerly direction through agricultural and viticultural land, west of the coastal village of Haumoana in Central Hawke's Bay (Fig.7.46). The downstream sub-site lies 0.33km from the open ocean and <50m from the main Tukituki River channel. The rural upstream sub-site runs parallel to Haumoana Road, before the stream continues for an additional ~4.2km.

Access to the upstream sub-site did not require any landowner permissions or the use of the HBRC master key. Privately-owned horses used the land around the sub-site to graze, so it was imperative that care was taken whilst manoeuvring around the animals, whilst ensuring that noise levels were

kept to a minimum. The HBRC master key was required to gain vehicle access to the downstream sub-site which ran through a publicly accessible cycle path. Caution was essential whilst driving along the cycle path if cyclists or pedestrians were within a close proximity. Regular checking of the nets along both sub-sites ensured that security remained uncompromised.

<u>Drainage structure present:</u>

There are no drainage structures present along Grange Creek, therefore this site will be used as a control.

<u>Upstream</u>

GPS Coordinates (DD): -39.607292, 176.940008

Figure 7.47. The upstream habitat along Grange Creek. **A**: Heavily mown grass spans the width of the left bank. **B**: Partially submerged flaxes are scattered amongst heavily grazed grass along the right bank. **C**: Scouring at the water line has caused the active erosion of ~75% of each bank. **D**: Large quantities of woody debris have fallen into the water

The upstream tidal sub-site along Grange Creek has very little hydraulic heterogeneity as this backwater region of the stream has a very slow water flow with little turbulence and is without any fast-flowing components like riffles or waterfalls. The riparian vegetation along the left bank lacks diversity and maturity as heavily mown grass spans the ~2m width, reducing the amount of fish spawning habitat available within the stream (Fig.7.47A). The left bank has an average height of ~1m from the water's surface, which casts shading across ~25% of the waterway, inhibiting any excessive macrophyte growth along the bank's edge and thus limiting the amount of suitable fish cover. Partially submerged flaxes are scattered amongst heavily grazed grass along the ~2m wide right bank, providing additional resting, feeding or spawning habitat for migrating fish (Fig.7.47B). Isolated

patches of water discolouration within the stream have been caused by the defecation by livestock who are known to regularly graze upon the right bank.

Although there is little turbulence along the stream, ~75% of each bank shows signs of active erosion. The daily fluctuations of the tide have led to significant scouring at the water line, and the subsequent undercutting of the banks which have released small substrates like cobbles and plant matter into the water channel (Fig.7.47C). At least 75% of the stream bed is covered by fine sediment like silt and sand, though conditions are unfavourable for any substantial EPT invertebrate colonisation as macrophyte growth remains limited. Considerable quantities of large woody debris have fallen from the right bank into the stream, often spanning the width of the waterway (Fig.7.47D). This offers an abundance of additional shelter for aquatic species, though excessive volumes may act as potential barriers for migrating fish.

<u>Downstream</u>

Note: Refer to the 'Downstream' section in the Memorial Park Drain site description for the sub-site GPS coordinates, site images and full habitat description.

Results

<u>Inanga and All Species Total Abundance</u>

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

<u>Model Justification:</u>

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 601.10) yields a Pearson-chisq dispersion value of 69.56 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 99.25) yields a Pearson-chisq value of 1.42 (p = 0.17) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.73 (p = 0.23) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 3723.54) yields a Pearson-chisq dispersion value of 103.99 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 332.30) yields a Pearson-chisq statistic of 0.83 (p = 0.91) = overdispersion does not exist. An additional Vuong AIC z statistic of 6.12E+4 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

<u>Results:</u>

The inanga abundance in the upstream (US) sub-site (N = 462) is significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 28) along Grange Creek (p = 0.0008) (Fig.7.48). An IRR value of 16.50 for the US sub-site (95% CI: 3.19 – 85.41) is considerably higher than an IRR value

of 4.67 for the DS sub-site (95% CI: 1.42 – 15.34), showing that the abundance of inanga is higher upstream and that upstream migrations are not restricted along Grange Creek.

The total abundance of all species in the US sub-site (N = 468) is also higher than the total abundance of all species in the DS sub-site (N = 279) along Grange Creek. However, an IRR value of 1.88 for the US sub-site (95% CI: 0.40 – 8.85) is not higher than an IRR value of 4.15 for the DS sub-site (95% CI: 1.39 – 12.43) and the difference is therefore not significant (p = 0.43) (Fig.7.48).
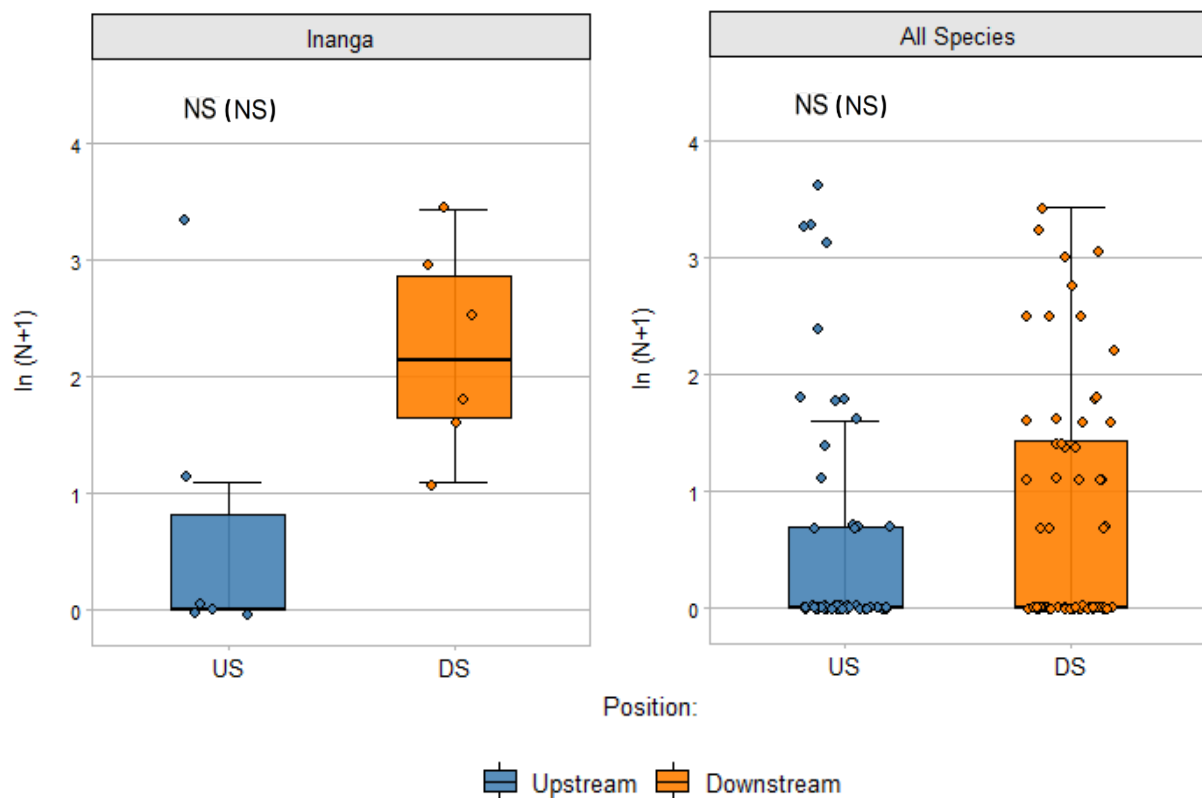

Figure 7.48. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Grange Creek site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and 'All Species').

Species Diversity and Similarity

H_0: *Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.*

H_1: *Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.*

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.07) along Grange Creek is significantly different to the H' value of the DS sub-site (H' = 0.68) as an US ENS value of 1.08 (H'_{max} = 1.10, p = 6.60E-15) and a DS ENS value of 1.96 (H'_{max} = 1.79) shows that a greater number of species inhabit the downstream reaches of the waterway, despite no drainage structure being present between the two sub-sites.

Pielou's evenness index (J') and species richness (S) are lower upstream (J' = 0.07, S = 3) than they are downstream (J' = 0.38, S = 6), though a Sørensen's similarity index of 0.67 indicates that 67% of species present across the site are found within both sub-sites. In fact, the migratory inanga and the resident common bully and crab are the only three species present within both of the US and the DS sub-sites.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature is similar within both sub-sites, although the dissolved oxygen saturation and the specific conductivity is higher downstream. As there is no barrier structure present along the stream, the habitat conditions of the downstream sub-site alone have led to the higher observed levels of DO and SPC. A higher macrophyte density along the downstream sub-site may have led to the increase in DO, whilst the highly tidal nature and the close proximity of the downstream sub-site to the open ocean has led to the high observed SPC levels across the sub-site, providing the optimal conditions for a higher fish biodiversity.

7.2.04. Te Ngarue Stream (tnst)

Site Description

Date of Study:

Wednesday 22nd January 2020.

Weather Conditions:

High sunshine levels with increasing cloud cover in the afternoon; warm and stable air temperature throughout the day.

Hydrographic Tide Predictions:

Tidal site. Lowest tide of 0.3m at 10:26 rising to 1.6m by 16:33 (LINZ, 2020).
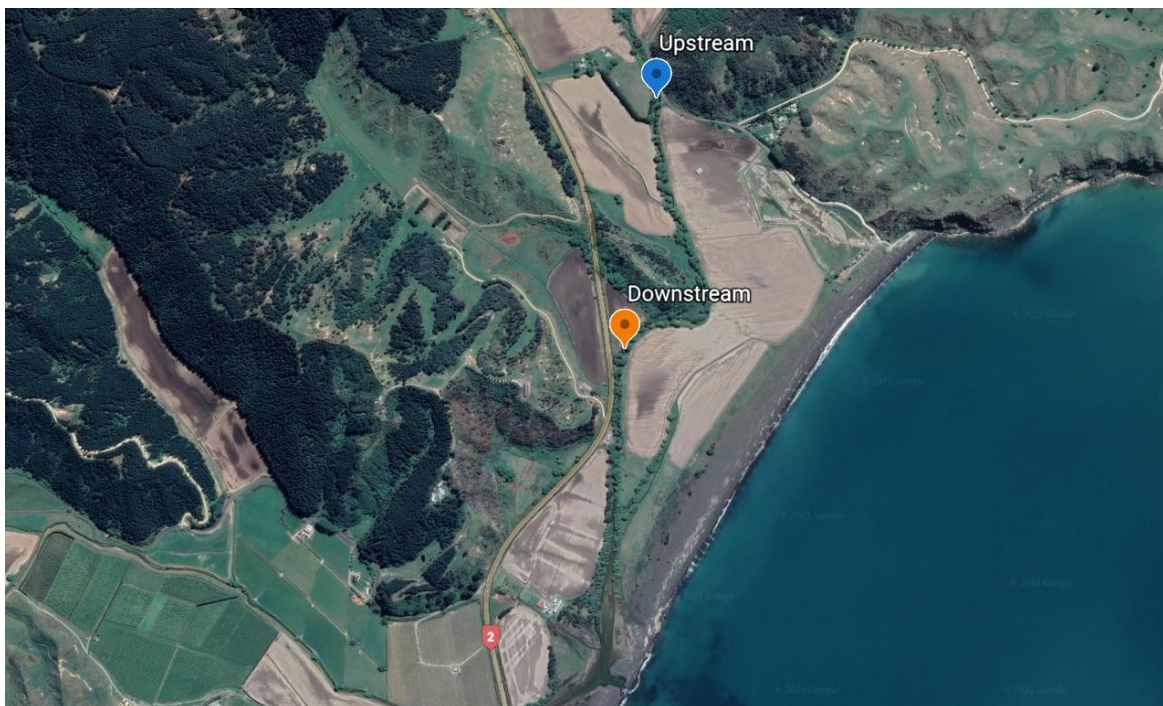
Figure 7.49. Google Earth image showing the location of the studied upstream and downstream Te Ngarue Stream sub-sites in relation to the Tangoio Forest, State Highway 2 and the open ocean (Google Earth, 2021).

Sharing a mouth with the Pakuratahi Stream, the Te Ngarue runs in a northerly direction through rural agricultural land in Northern Hawke's Bay (Fig.7.49). The downstream sub-site lies ~1km from the open ocean and runs parallel to State Highway 2, east of the Tangoio forest. The stream meanders east, before continuing north towards a dirt track road which is regularly used to access the nearby Tangoio beach. The Te Ngarue Stream continues above the upstream sub-site for >6.2km, though a number of tributaries branching in all directions from the stream may significantly increase the amount of available upstream habitat.

Both sub-sites ran through privately-owned land, therefore access required the permissions of local landowners who were contacted using the HBRC phone book. As the public did not have access to either sub-site, net security remained good throughout the day.

There are no drainage structures present along the Te Ngarue Stream, therefore this site will be used as a control.

Upstream

GPS Coordinates (DD): -39.333387, 176.914663

Figure 7.50. The upstream habitat along the Te Ngarue Stream. **A**: Shading is cast from the left and the right banks across ~60% of the stream. **B**: Mature native trees with a diverse understorey span both banks. **C**: Significant bank slumping has led to the exposure of large root mats. **D**: Turbulent riffles and a small waterfall provide favourable conditions for EPT colonisation.

The upstream sub-site along the Te Ngarue Stream has significant hydraulic heterogeneity as there are riffles and waterfalls with a fast flow and turbulence along the upper and lower regions of the stream, whilst deep pools with a slow water flow are scattered across the centre of the sub-site. Mature native trees line the upper regions of both the left and the right banks, forming a dense, overhanging canopy which casts shading across ~60% of the stream (Fig.7.50A). Diverse shrubs, grasses and aquatic plants span the ~5m wide understorey along each bank, encroaching into the stream and providing an abundance of favourable habitat for migrating fish to rest, seek refuge or spawn (Fig.7.50B). Each bank has a height of 2-3m from the water's surface, casting further shading across the stream, although areas with sparse overhanging riparian vegetation allows light to diffuse into the waterway. As the stream is exposed to a limited amount of direct sunlight, macrophyte growth across the sub-site is minimal, therefore the amount of suitable shelter for aquatic species is reduced.

Approximately 50% of each bank shows signs of significant recent erosion, mainly due to bank slumping caused by the large overhanging riparian vegetation. The eroding banks have led to the

release of woody debris, plant matter and small substrates like cobbles and silt directly into the water channel, offering further potential fish shelter habitat. The erosion has formed undercut banks and has subsequently led to the exposure of large root mats, which may act as additional fish spawning habitat if partially submerged within deep pools (Fig.7.50C). Small boulders and cobbles, clear of filamentous algae and macrophytes cover >75% of the stream bed which, during a low tide, increases the flow of water downstream and provides an abundance of suitable habitat for EPT invertebrate colonisation. These riffles, along with a small, unobstructive waterfall within the stream offer the favourable conditions required by migrating fish to continue their upstream journey (Fig.7.50D).

Downstream

GPS Coordinates (DD): -39.339692, 176.913698

Figure 7.51. The downstream habitat along the Te Ngarue Stream. **A**: Mature, native trees are scattered amongst natural grasses along the left bank. **B**: Young trees form a dense canopy above the right bank. **C**: Shading is cast across ~70% of the waterway, though some light penetrates through the tree canopy. **D**: Fallen woody debris provides an abundance of additional fish shelter.

The downstream sub-site along the Te Ngarue Stream has limited hydraulic heterogeneity compared with the upstream sub-site, as the stagnant backwater has little turbulence and a very slow water flow. Mature, native trees are scattered along the ~5m wide left bank, amongst a riparian strip of mainly short natural grasses and regenerating shrubs (Fig.7.51A). Young trees span the ~7m width of the right bank, forming a dense canopy with a bare and impervious understorey, though long grasses and young shrubs have grown in isolated patches where trees have fallen from the bank (Fig.7.51B). Shading is cast across ~70% of the waterway from both banks, though the stream remains exposed to direct sunlight along areas where overhanging vegetation is sparse or where there are large

enough gaps within the tree canopy to allow light to penetrate through (Fig.7.51C). However, macrophyte growth is minimal across both shaded and non-shaded areas of the stream, limiting the amount of available shelter for aquatic species.

As with the upstream sub-site, approximately 50% of each bank shows signs of recent erosion as the large overhanging trees on each bank have caused significant bank slumping. Small substrates like cobbles and plant matter have been released into the water channel and have settled along the stream bed, which fine sediment covers at least 75%. A high volume of woody debris and plant matter, fallen from the right bank has accumulated along the water's edge, causing some slight turbulence further downstream (Fig.7.51D). The patches of woody debris have provided a copious amount of suitable fish resting and spawning habitat, though conditions remain unfavourable for EPT invertebrate colonisation.

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A suitable, well-fitting Poisson GLM (AIC = 18.77) yields a Pearson-chisq dispersion value of 0.4 (p = 0.95) = overdispersion does not exist. An additional Vuong AIC z statistic of 1.85E+05 (p < 2.2E-16) shows that the **Poisson GLM** is favoured over a zero-inflated Poisson GLM (AIC = 22.77) as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 739.13) yields a Pearson-chisq dispersion value of 12.07 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 247.73) yields a Pearson-chisq statistic of 0.66 (p = 1) = overdispersion does not exist. An additional Vuong AIC z statistic of 2.53 (p = 0) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 6) is higher than the inanga abundance in the downstream (DS) sub-site (N = 0) along the Te Ngarue stream, although the difference is not statistically significant (p = 1) (Fig.7.52), showing that the abundance of inanga is marginally higher upstream and that upstream migrations are unrestricted. An US IRR value of infinite arose (95% CI: 0 – infinite) as a statistical comparison against the downstream reference group with a sample size of 0 is not possible. The small sample sizes associated with each sub-site also led to the IRR's wide CIs.

The total abundance of all species in the US sub-site (N = 121) is higher than the total abundance of all species in the DS sub-site (N = 39) along the Te Ngarue stream, although the difference is also not statistically significant (p = 0.10) (Fig.7.52). An IRR value of 3.10 for the US sub-site (95% CI: 0.82 –

11.76) is higher than an IRR value of 0.65 for the DS sub-site (95% CI: 0.25 – 1.70), showing that a higher total abundance of all species inhabit the upstream reaches of the stream, and that upstream passage is unrestricted.
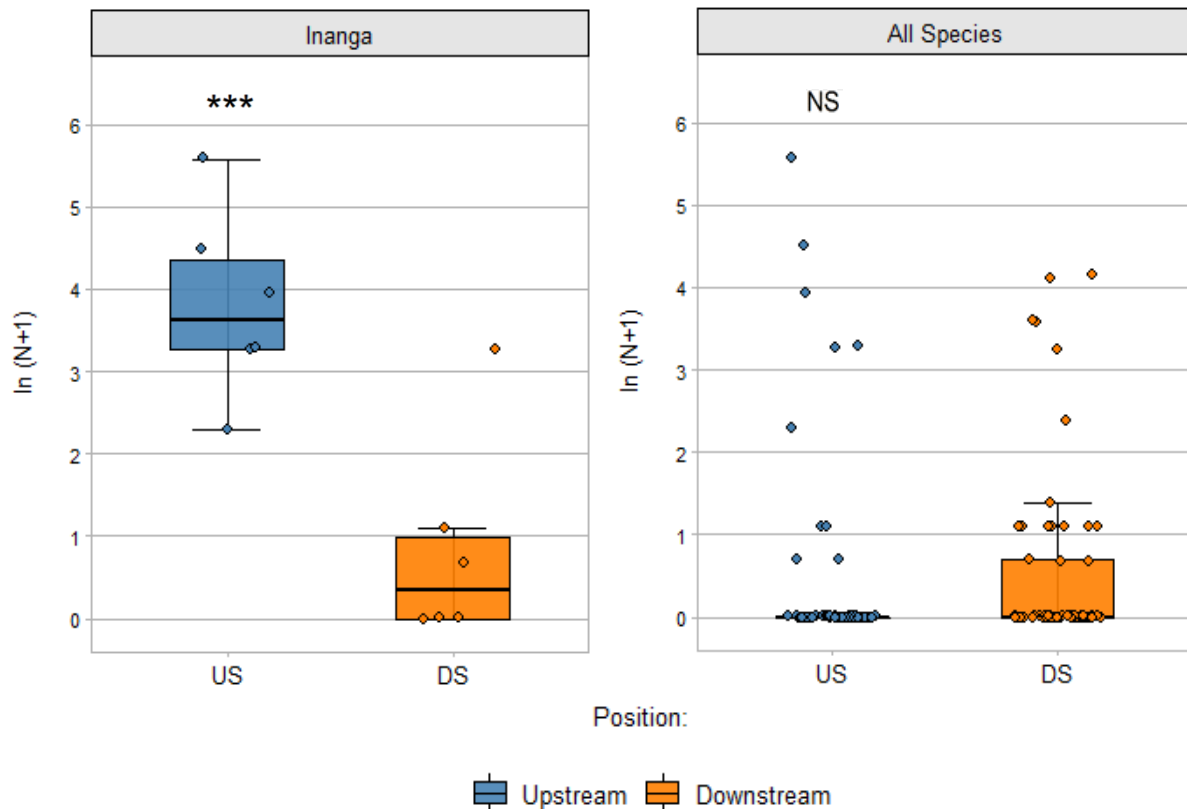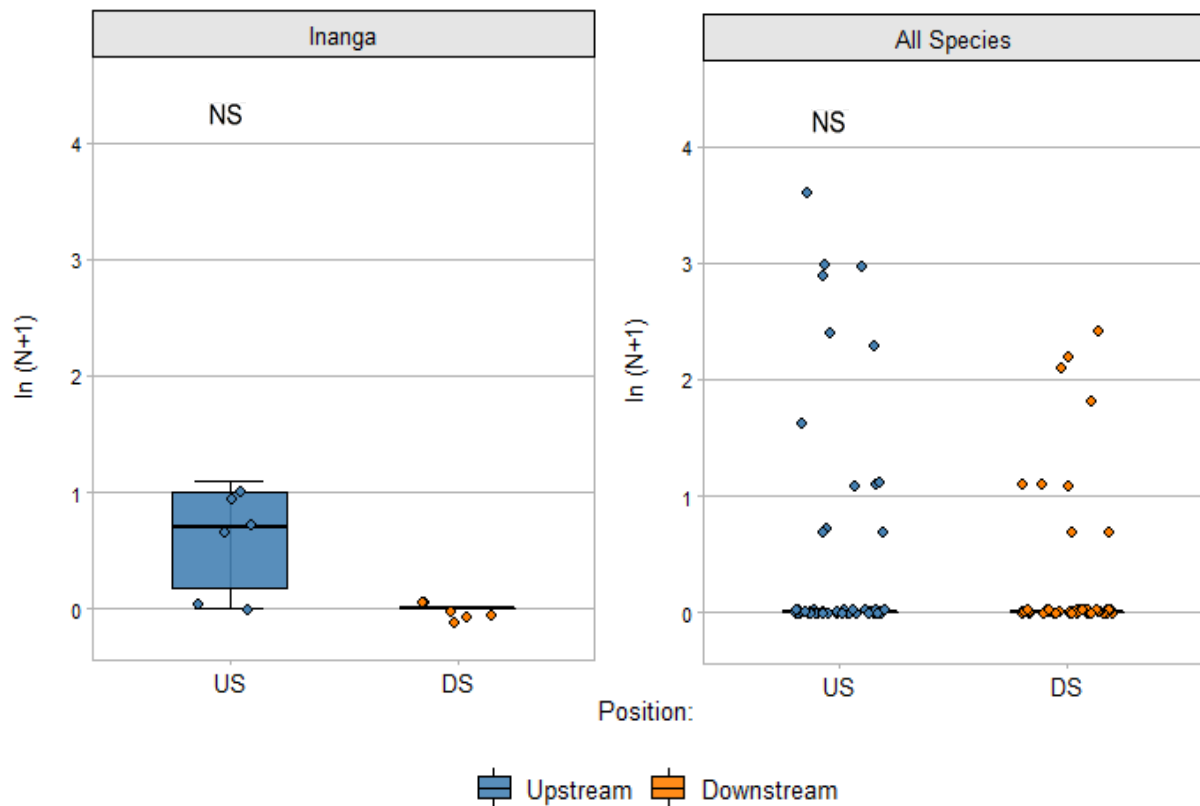

Figure 7.52. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Te Ngarue Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (Poisson GLM for 'Inanga' and negative binomial GLM for 'All Species').

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.45) along the Te Ngarue stream is not significantly different to the H' value of the DS sub-site (H' = 0.70) as an US ENS value of 1.58 (H'_{max} = 1.39, p = 0.16) and a DS ENS value of 2.01 (H'_{max} = 1.39) shows that a similar number of species inhabit both of the upstream and the downstream sub-sites and that upstream passage is unrestricted.

Pielou's evenness index (J') is slightly lower upstream (J' = 0.33) than it is downstream (J' = 0.50), although the species richness (S) is the same within both sub-sites (S = 4 for both). A Sørensen's similarity index of 0.25 indicates that only 25% of species present across the stream are found within both sub-sites. The invasive mosquitofish is the only species present within both of the US and the

DS sub-sites, whereas the migratory inanga and the resident common bully are present within the US sub-site only.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean specific conductivity is similar within both sub-sites, although the temperature and the dissolved oxygen saturation differs slightly. As there is no barrier structure present along the stream, the habitat conditions of the upstream sub-site alone have led to the slightly higher observed temperature and DO levels. Unlike the downstream sub-site which consists only of slow flowing deep pools, turbulent riffles along the upstream sub-site may have led to increased hydraulic and nutrient mixing, leading to increased DO saturation across the stream. The shallower riffles are also more exposed to direct sunlight, leading to increased water temperatures and providing suitable conditions for inanga to inhabit or migrate through.

7.2.05. Pakuratahi Stream (past)

Site Description

Date of Study:

Thursday 23rd January 2020.

Weather Conditions:

High sunshine levels with no cloud cover throughout the day; high air temperatures with a slight ocean breeze at the downstream site.

Hydrographic Tide Predictions:

Tidal site. Lowest tide of 0.3m at 11:21 rising to 1.6m by 17:27 (LINZ, 2020).

Figure 7.53. Google Earth image showing the location of the studied upstream, midstream and downstream Pakuratahi Stream sub-sites in relation to the Tangoio forest, State Highway 2 and the open ocean (Google Earth, 2021).

The Pakuratahi Stream runs in a westerly direction through agricultural land, towards the Tangoio forest in Northern Hawke's Bay (Fig.7.53). The downstream sub-site lies 0.4km west from the open ocean and runs beneath a roadway bridge which holds the SH2 (New Zealand's State Highway 2). The stream continues west, running parallel to the rural Pakuratahi Valley road, which eventually becomes a narrow, unnamed dirt track that runs through the stream, and divides the upstream from the midstream sub-site. The midstream sub-site lies directly below the dirt track, whilst the upstream lies directly above. The Pakuratahi Stream continues above the upstream sub-site for ~2km, though a considerable number of tributaries branch from the stream in all directions through the Tangoio forest, so the true amount of available upstream habitat may be significantly higher.

Neither landowner permissions, nor the use of the HBRC master key were required to gain access to the Pakuratahi Stream sub-sites, as all ran through publicly accessible land. Site security remained good throughout the day as the sites are not commonly used by the public.

Figure 7.54. A ford divides the upstream and midstream sub-sites along the Pakuratahi Stream. **A**: The ford's drop causes may act as a potential barrier to upstream migrating fish. **B**: The ford crosses a dirt road which is used by heavy logging trucks.

There are no drainage structures present along the Pakuratahi Stream, therefore this site will be used as a control. However, a ford measuring 7m in length and 6m in width runs over a dirt track which divides the upstream and midstream sub-sites (Fig.7.54). Although not a drainage structure, the ford may still act as a potential barrier to fish migrations and will therefore be assessed for fish passability.

Upstream

GPS Coordinates (DD): -39.347163, 176.876969
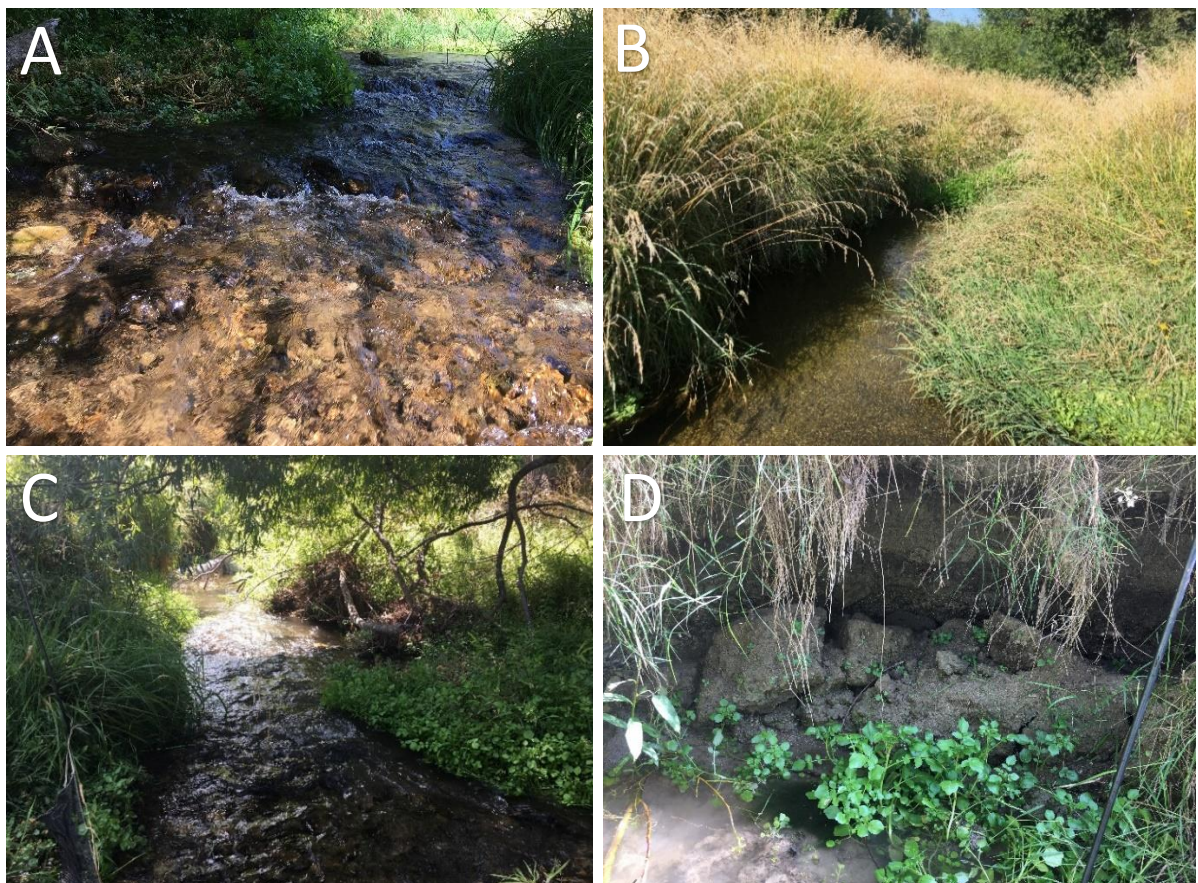
146

The upstream sub-site along the Pakuratahi stream has numerous hydraulic components, offering a diverse range of suitable habitats for aquatic species to feed, shelter and spawn within. A riffle along the centre of the sub-site causes there to be a fast flow of water to travel downstream, increasing the amount of turbulence across the site (Fig.7.55A). Fast-flowing water over cobbles and small boulders which are free of any macrophytes or filamentous algae, provide the favourable conditions for EPT invertebrate colonisation, whilst also allowing a suitable environment for small-bodied fish to continue their upstream migrations. Directly above and below the riffle, deep pools with a slow water flow and little turbulence line each bank's edge. A dense riparian strip of long, overhanging grasses span each of the ~5m wide left and right banks, which encroaches into the stream's deep pools and provides an abundance of suitable resting and spawning habitat for migrating fish (Fig.7.55B). The dense, overhanging bank vegetation, as well as a single, large mature tree along the right bank, which has a diverse and intact understorey, casts shading across ~70% of the stream, preventing any excessive growth of submerged or floating macrophytes across the sub-site (Fig.7.55C).

The turbulence caused by the fast-flowing water over the riffle, and the slumping of each bank caused by the encroaching riparian vegetation has led to the active or recent erosion of ~35% of each bank. Undercut banks in some areas have caused minor landslips which have released fine sediments, like silt and soil, woody debris and plant matter into the water channel (Fig.7.55D). However, root mats exposed by eroding banks and fallen substrates within the stream have been utilised by invertebrate and fish species as additional shelter and feeding habitat.

Midstream

GPS Coordinates (DD): -39.346927, 176.878214
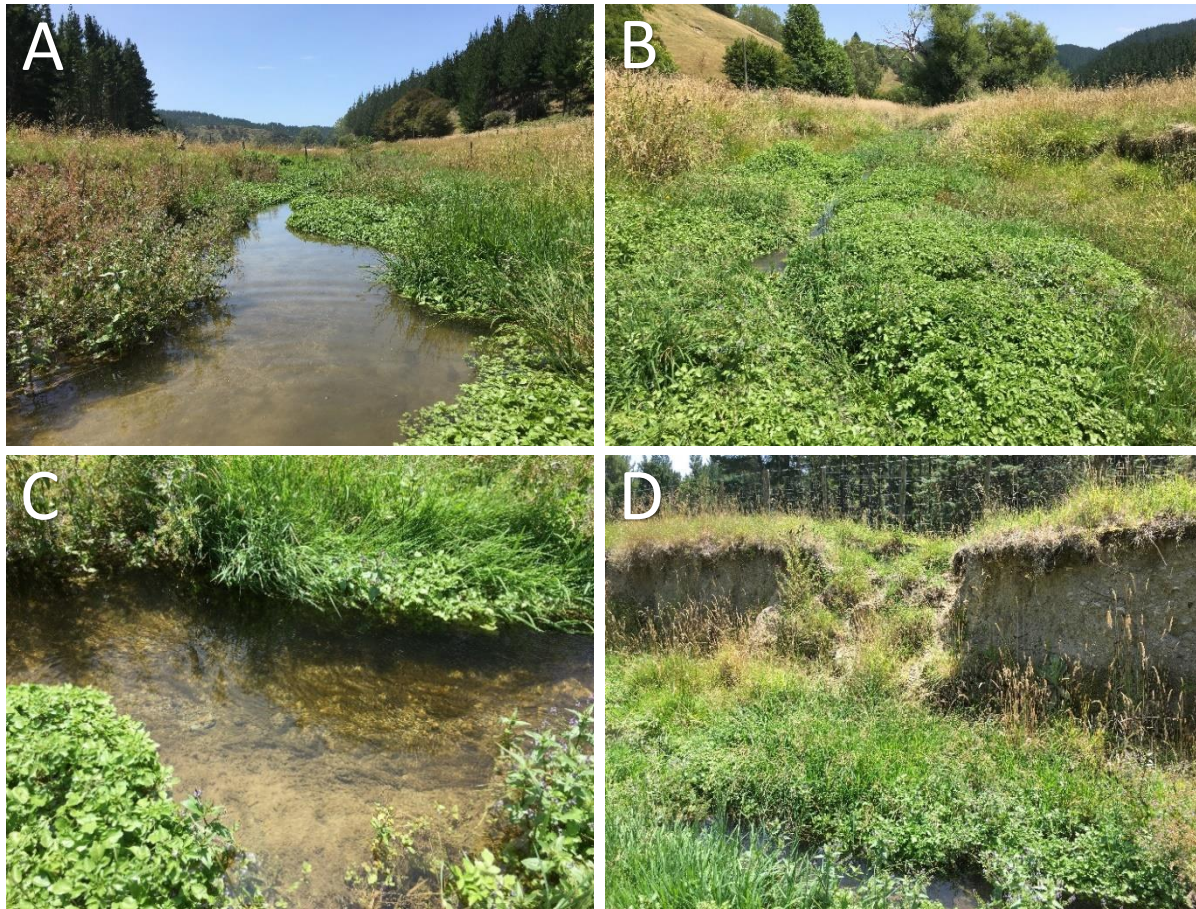
Figure 7.56. The midstream habitat along the Pakuratahi Stream. **A**: Long grasses and young shrubs span both of the left and the right banks. **B**: Excessive growth of submerged and emergent macrophytes choke the water channel. **C**: Only a narrow channel free of macrophytes remains for migrating fish. **D**: Slumping banks and subsequent landslips have been caused by active erosion.

The midstream sub-site along the Pakuratahi stream has some hydraulic heterogeneity, though there are no lotic components, nor any turbulence compared with the upstream sub-site. The midstream region is mainly a backwater with pools of varying depths and a slow water flow. As with the upstream sub-site, long exotic grasses and young shrubs span each of the ~7m wide left and right banks, encroaching far into the stream and offering copious amounts of suitable fish spawning habitat (Fig.7.56A). Due to the lack of tree cover and mature, overhanging shrubs along either bank, shading is cast across only 5-10% of the stream at any period of the day. This has resulted in the excessive growth of submerged and emergent macrophytes, which choke ~75% of the water channel (Fig.7.56B). Although these dense aquatic plants offer a substantial volume of favourable fish cover and resting habitat, the extremely high abundance has led to significant regions of the stream becoming obstructed, leaving only a narrow channel for migrating fish to easily navigate through (Fig.7.56C).

The level of active or recent erosion along the midstream sub-site is significantly higher than that along the upstream sub-site; approximately 75% of each bank shows signs of substantial degradation. The fluctuating water levels along the tidal site during flood events have caused scouring at the water line, which has led to the undercutting of the banks and subsequent major landslips (Fig.7.56D). Large quantities of fine sediment, woody debris and plant matter have fallen

into the stream, becoming either entangled within the dense macrophyte layers or resting beneath the water line. The fallen substrates cover >75% of the stream bed, offering additional shelter and refuge habitat for fish and invertebrate species.

Downstream

GPS Coordinates (DD): -39.349476, 176.909491

Figure 7.57. The downstream habitat along the Pakuratahi Stream. **A**: Mature shrubs and long grasses span the width of the left bank. **B**: Long grasses and dense bramble bushes encroach into the stream from the right bank. **C**: Mature, overhanging trees line the lower regions of each bank. **D**: A road bridge casts significant shading across the width of the

The tidal downstream sub-site along the Pakuratahi stream has little hydraulic heterogeneity compared to the upstream and the midstream sub-sites, although during a high tide, there is a strong flow of water with some turbulence across the width of the stream. Mature shrubs, long exotic grasses and dense bramble bushes span the upper regions of the ~5m wide left and right banks, encroaching into the stream above the road bridge, and providing an abundance of suitable fish resting and spawning habitat (Fig.7.57A, Fig.7.57B). Large mature, overhanging trees with an intact understorey span the lower regions of each bank, casting shading across ~15% of the stream below the road bridge (Fig.7.57C). The road bridge itself spans the width of the stream and casts a significant amount of shading across a 20mx10m area of the waterway (Fig.7.57D). Macrophyte growth is minimal along both shaded and non-shaded areas of the stream, reducing the amount of fish cover and resting habitat available across the sub-site.

Although the stream's encroaching riparian vegetation offers some protection against scouring at the water line, approximately 25% of each bank shows signs of active erosion. The daily tidal fluctuations, fast flow and high turbulence have led to the undercutting of some areas of the bank,

causing small substrates like silt and cobbles to fall into the stream. At least 75% of the stream bed is covered by fine sediment like silt and sand, though only ~15% is suitable for EPT invertebrate colonisation. Woody debris and plant matter released from the riparian vegetation are scattered in isolated patches along each bank, which become exposed as the water level recedes during a low tide. The water flow slows within the temporary wetland habitats, creating additional suitable refuge or spawning grounds for migrating fish.

Results

Note: Due to the small length and the limited space along the upstream sub-site, only four fyke nets were set. Only four fyke nets were also set within the downstream sub-site as water levels were too deep in some areas for efficient net setting.

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream, nor midstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream, and midstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 158.50) yields a Pearson-chisq dispersion value of 9.27 ($p < 2.2E-16$) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 91.93) yields a Pearson-chisq value of 0.94 ($p = 0.50$) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.23 ($p = 0.41$) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 1486.83) yields a Pearson-chisq dispersion value of 19.72 ($p < 2.2E-16$) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 297.51) yields a Pearson-chisq statistic of 0.85 ($p = 0.90$) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.14 ($p = 0.44$) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 41) is not higher than the inanga abundance in the downstream (DS) sub-site (N = 149) along the Pakuratahi stream as an IRR value of 0.28 for the US sub-site (95% CI: 0.08 – 0.94) is not higher than an IRR value of 37.25 for the DS sub-site (95% CI: 15.89 – 87.34), although the difference is statistically significant ($p = 0.04$) (Fig.7.58). The inanga abundance in the midstream (MS) sub-site (N = 9) is also not higher than the inanga abundance in the DS sub-site along the Pakuratahi stream as an IRR value of 0.04 for the MS sub-site (95% CI: 0.01 – 0.14) is not higher than the DS IRR value, although the difference is also statistically significant ($p = 7.56E-07$) (Fig.7.58). This shows that although the abundance of inanga is highest downstream, inanga are still able to migrate into the upstream and midstream sub-sites and that upstream migrations are therefore unrestricted.

The total abundance of all species in the US sub-site (N = 43) is not higher than the total abundance of all species in the DS sub-site (N = 357) along the Pakuratahi stream as an IRR value of 0.12 for the US sub-site (95% CI: 0.03 – 0.58) is not higher than an IRR value of 8.93 for the DS sub-site (95% CI: 3.00 – 26.55), although the difference is statistically significant (p = 0.06) (Fig.7.58). The total abundance of all species in the MS sub-site (N = 10) is not higher than the total abundance of all species in the DS sub-site along the Pakuratahi stream as an IRR value of 0.02 for the MS sub-site (95% CI: 0 – 0.09) is not higher than the DS IRR value, although the difference is also statistically significant (p = 3.76E-07) (Fig.7.58). As with the inanga abundance analysis, these findings show that although the total abundance of all species is higher within the downstream sub-site, fish are still able to navigate into the upstream and the midstream sub-sites and that upstream passage is therefore unrestricted.
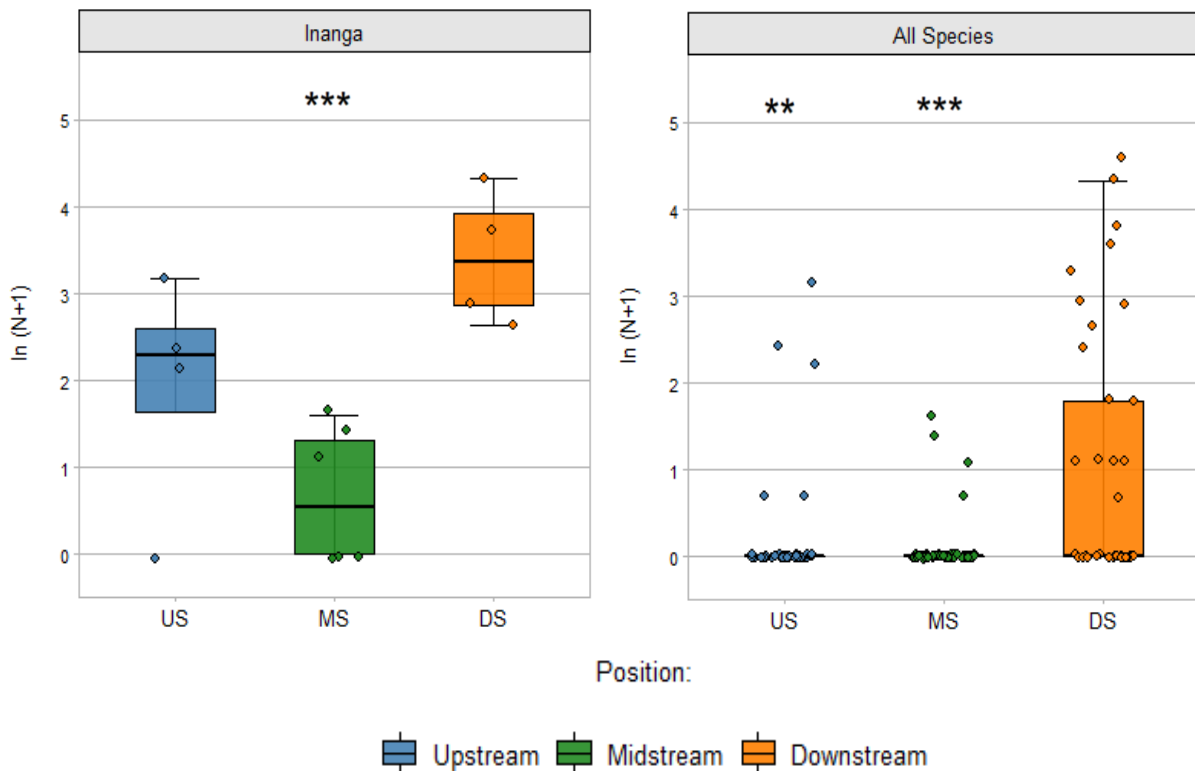
Figure 7.58. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US), midstream (MS) and downstream (DS) positions along the Pakuratahi Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and 'All Species').

Species Diversity and Similarity

H_0: *Biodiversity, species richness and species evenness upstream, and midstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.*

H_1: *Biodiversity, species richness and species evenness upstream, and midstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.*

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.22) and the MS sub-site (H' = 0.33) along the Pakuratahi stream is significantly different to the H' value of the DS sub-site (H' = 1.23) as an US ENS value of 1.25 (H'$_{max}$ = 1.10, p = 1.90E-10) and a MS ENS value of 1.38 (H'$_{max}$ = 0.69, p = 2.33E-03) is considerably lower than the DS ENS value of 3.42 (H'$_{max}$ = 1.95), showing that there is a greater biodiversity downstream than there is upstream or midstream. This was expected however as the downstream sub-site lies along the freshwater-saltwater boundary zone and is within close proximity to the open ocean; an ecotone which is typically associated with a greater biodiversity than fully freshwater habitats.

Pielou's evenness index (J') and species richness (S) were lower in the US sub-site (J' = 0.20, S = 3) and in the MS sub-site (J' = 0.47, S = 2) than they are within the DS sub-site (J' = 0.63, S = 7). A Sørensen's similarity index (ß) of 0.40 indicates that only 40% of species present across the site are found within both of the US and the DS sub-sites; the migratory inanga and the resident common bully are the only two species present within both sub-sites. A ß value of 0.44 indicates that only 44% of species are present within both of the MS and the DS sub-sites; the two shared species being the migratory inanga and crab.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream, the midstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, the dissolved oxygen saturation and the specific conductivity measures are similar within both of the upstream and the midstream sub-sites, but are all higher within the downstream sub-site. As there is no barrier structure present along the stream, the habitat conditions of each sub-site alone have led to the differences in water qualities. The lack of shading downstream and the close geographical proximity to the open ocean has led to increased water temperatures and higher SPC levels, whereas the higher DO saturation may be caused by the highly tidal nature and the faster flow velocity of the downstream sub-site as tidal exchange is greater than that across the upstream and the midstream sub-sites, conditions which inanga have thrived within in abundance.

7.2.06. Mangarau Stream (mgst)

Site Description

Date of Study:

Thursday 19th December 2019.

Weather Conditions:

High levels of sunshine with light cloud cover; warm and stable air temperature throughout the day.

Hydrographic Tide Predictions:

Non tidal site. Lowest tide of 0.2m at 05:52 rising to 1.8m by 12:08 (LINZ, 2020).
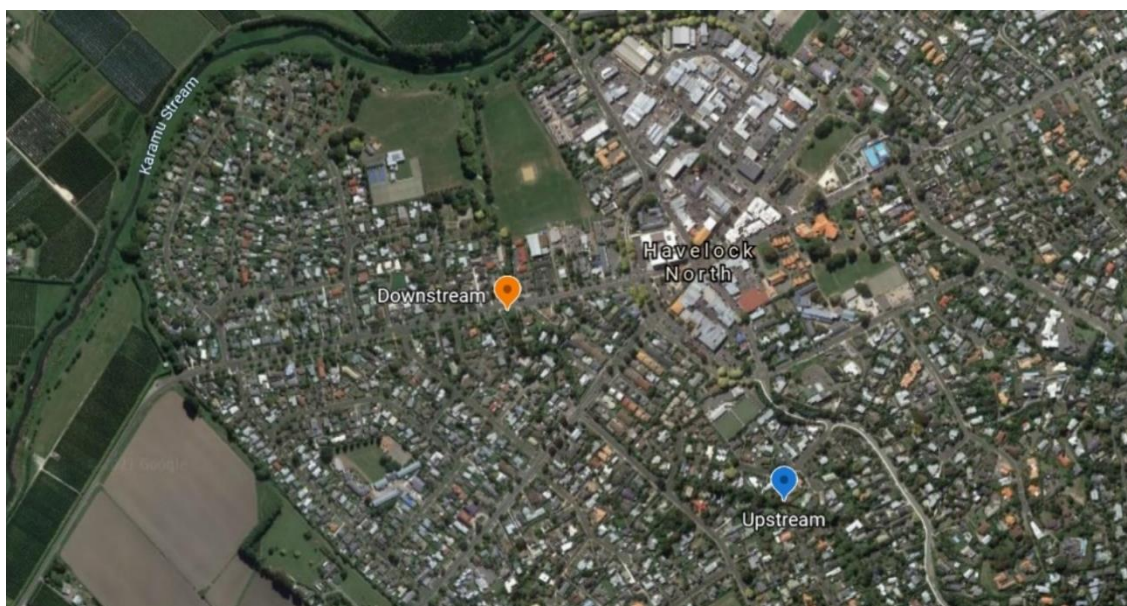
Figure 7.59. Google Earth image showing the location of the studied upstream and downstream Mangarau Stream sites in relation to the Karamu Stream and the urban town of Havelock North (Google Earth, 2021).

Branching from the Karamu Stream, which itself is a tributary of the Clive River, the Mangarau Stream runs in a southerly direction through the town of Havelock North in Southern Hawke's Bay (Fig.7.59). Situated by the Anderson public park, the downstream sub-site lies ~19.4km from the open ocean and runs beneath the busy Te Aute Road bridge. The stream continues in a south-easterly direction, through the urban upstream sub-site where private gardens line each of the stream's banks. Habitat upstream continues for >7.8km through Havelock North and the Tainui Reserve before branching into several smaller streams in the Kohinurakau Hill Range.

Both sub-sites ran through public areas, therefore access was good and permissions or use of a HBRC master key were not required. Regular checking of the nets throughout the day ensured that the sites' security remained sufficient.

Drainage structure present:

Figure 7.60. A bridge apron divides the upstream and downstream sub-sites along the Mangarau Stream. **A**: Baffles with alternately placed gaps span the length of the structure. **B**: A drop on the lower end of the bridge apron may act as a fish barrier.

A bridge apron spanning 190m in length divides the Mangarau Stream's upstream and downstream sub-sites and runs beneath one of Havelock North's busiest roadways. Baffles with alternately placed gaps span the bridge apron with the aim of reducing water flow and retaining a passageway upstream for migrating fish (Fig.7.60A). However, at the most downstream point of the bridge apron, a single drop ledge spans the width of the stream, acting as a potential migratory barrier (Fig.7.60B). The ledge spans ~4.5m in length and has a height of ~0.7m from the stream bed, though the drop during periods of low rainfall is ~0.3m.

Upstream

GPS Coordinates (DD): -39.674164, 176.87899

The Mangarau Stream's upstream reach has high habitat heterogeneity as a variety of hydraulic components span the length of the sub-site. Between 4 and 5m of diverse and natural riparian vegetation spans the left and the right banks which are constrained on each side by high rising garden fences. Dense shrubs and common ivy *(Hedera helix)* line each bank and encroach into the stream below (Fig.7.61A), providing suitable cover for aquatic species. Large overhanging canopy trees are scattered along each of the banks, casting shade across large areas of the stream and decreasing the level of macrophyte growth within the waterway (Fig.7.61B). Although sparse, macrophyte growth still occurs in minor quantities when sunlight diffuses through the canopy, providing additional resting sites for fish. A recent landslip beneath an overhanging tree suggests active bank slumping and erosion which has resulted in the release of root mats and woody debris into the waterway (Fig.7.61C).

Deep lentic pools with no turbulence are present along the lowest reaches and deposited fine sediment lines the stream bed. The water level significantly lowers further upstream and the diversity of substrate types increases. An abundance of boulders, cobbles, gravel and sand line the stream bed, increasing the water flow and turbulence downstream (Fig.7.61D). Approximately 70% of the substrate is clear of filamentous algae, providing favourable conditions for EPT invertebrate colonisation. Small pools with little turbulence have formed around the bank's edges, establishing suitable resting sites for migrating fish.

Downstream

GPS Coordinates (DD): -39.671219, 176.873164

Figure 7.62. The downstream habitat along the Mangarau Stream. **A & B**: Mature shrubs and young grasses span the width of the left and the right banks. **C**: Shading is cast beneath the road bridge which passes over the downstream sub-site. **D**: Boulders and cobbles along the shallowest areas of the stream bed increase the water flow velocity.

The riparian vegetation along the banks of the Mangarau Stream's downstream sub-site span between 7 and 10m in width and are constrained by garden fences. Mature shrubs and young grasses span the left and the right banks, reaching down to the water's edge and often encroaching into the stream (Fig.7.62A, Fig.7.62B). Approximately 25% of the stream bank is undergoing active erosion due to scouring at the water line where overhanging vegetation is not present. Shading is cast across the width of the stream in the upper reaches of the sub-site by large native trees, with additional shading being cast by a road bridge within the lower reaches (Fig.7.62C). Both banks beneath the road bridge span ~2m and have little vegetation, though some young, exotic grasses grow around the bridge's gaps. Macrophytes are sparse along the sub-site's shaded areas, though curly pondweed *(Potamogeton crispus)* is found in abundance where little or no shading is cast onto the stream, providing suitable cover and resting habitat for aquatic species.

As with the upstream, the downstream sub-site has high hydraulic heterogeneity as deep pools of slow flowing water are present along most of the waterway, though a turbulent riffle beneath the road bridge increases the water flow velocity along this stretch. Boulders, cobbles, sand and woody debris line the stream bed in the shallower areas, providing suitable conditions for EPT colonisation and resting sites for migrating fish (Fig.7.62D).

Results

Inanga and All Species Total Abundance

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 30.98) yields a Pearson-chisq dispersion value of 3.00 (p < 6.6E-04) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 23.87) yields a Pearson-chisq value of 0.58 (p = 0.83) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.27 (p = 0.39) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 260.74) yields a Pearson-chisq dispersion value of 7.13 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 120.21) yields a Pearson-chisq statistic of 0.69 (p = 1) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.09 (p = 0.47) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 1) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 5) along the Mangarau stream (p = 0.40) (Fig.7.63). An IRR value of 0.20 for the US sub-site (95% CI: 0.01 – 8.41) is not higher than an IRR value of 0.83 for the DS sub-site (95% CI: 0.08 – 8.61), showing that inanga abundances are higher downstream. However, as one inanga inhabits the upstream sub-site, upstream passage is possible beyond the bridge apron, and the structure therefore does not act as a full barrier to upstream migrations but acts as either a partial or a temporary barrier.

The total abundance of all species in the US sub-site (N = 15) is not significantly higher than the total abundance of all species in the DS sub-site (N = 25) along the Mangarau stream (p = 0.60) (Fig.7.63). The IRR value for the US sub-site is 0.60 times (95% CI: 0.09 – 4.05) larger than the DS sub-site IRR value of 0.42 (95% CI: 0.11 – 1.58), showing that a higher total abundance of all species inhabit the downstream reaches of the stream. However, a considerable population of fish still inhabit the upstream sub-site above the bridge apron and therefore the structure does not fully restrict the upstream passage of fish.
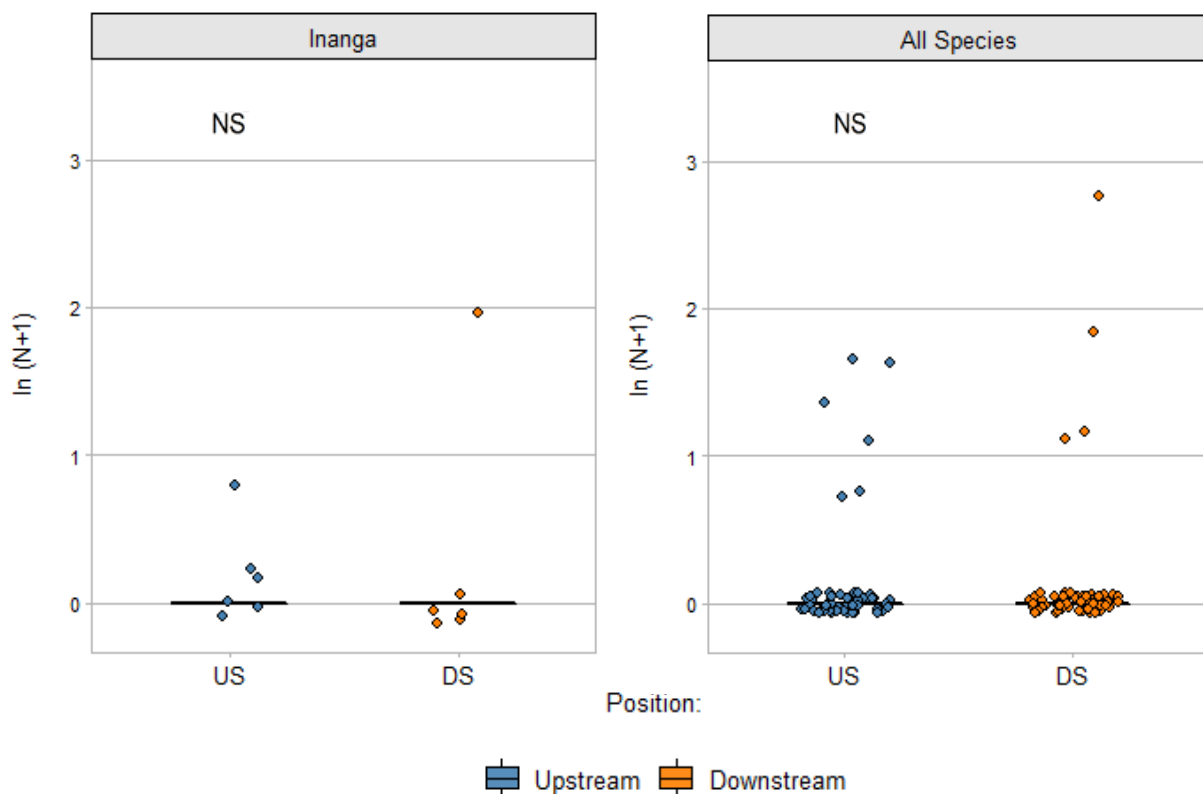
Figure 7.63. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Mangarau Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and 'All Species').

<u>Species Diversity and Similarity</u>

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.49) along the Mangarau Stream is not significantly different to the H' value of the DS sub-site (H' = 0.76) as an US ENS value of 1.62 (H'_{max} = 1.10, p = 0.33) and a DS ENS value of 2.14 (H'_{max} = 1.10) shows that there is a similar biodiversity of species within both sub-sites and that the bridge apron does not restrict upstream fish passage.

Pielou's evenness index (J') is lower in the US sub-site (J' = 0.44) compared with the evenness within the DS sub-site (J' = 0.69), although the species richness (S) US (S = 3) is the same as the richness DS (S = 3). A Sørensen's similarity index of 0.67 indicates that 67% of species present across the stream are found within both sub-sites as the migratory inanga and the resident common bully are the only two species present both upstream and downstream.

<u>Environmental Effect on Species Abundance</u>

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, the dissolved oxygen saturation and the specific conductivity measures are similar within both sub-sites, suggesting that the bridge apron structure does not act as an environmental barrier along the Mangarau stream. Similar habitats along each sub-site also suggests that the abundance of inanga is not affected by any environmental factors along the stream and that the physical bridge apron structure alone is the sole cause for any differences in inanga or total of all species abundances between sub-sites.

7.2.07. Ngaruroro Backwash (ngbw)

Site Description

Date of Study:

Tuesday 21st January 2020.

Weather Conditions:

Dense cloud cover in the morning with breaks of sunshine in the afternoon; stable and warm air temperature throughout the day.

Hydrographic Tide Predictions:

Tidal site. Lowest tide of 0.3m at 09:29 rising to 1.7m by 15:37 (LINZ, 2020).

Figure 7.64. Google Earth image showing the location of the studied upstream and downstream Ngaruroro Backwash sites in relation to the nearby Ngaruroro River, Highway 51 and the open ocean (Google Earth, 2021).

The Ngaruroro Backwash site runs parallel to the Ngaruroro River channel and is situated north of the village of Clive. The coastal highway, 51, runs perpendicular to the site and acts as one of the main roadways linking Hawke's Bay's two cities: Napier in the north and Hastings in the south (Fig.7.64). Access to the site requires the use of a HBRC master key to open a locked gate before a dirt track through agricultural land leads directly to the site's drainage structure.

The Ngaruroro Backwash runs directly from the Waitangi Estuary, the mouth of Muddy Creek, Ngaruroro, Tutaekuri and Clive Rivers. The habitat downstream of the site's drainage structure lies 1.32km from the open ocean in the east; habitat upstream of the potential barrier runs parallel to the Ngaruroro River for a further >1.7km in a south westerly direction. Multiple openings linking the Ngaruroro Backwash to another, similar sized waterway also running parallel to the main river, provide an additional 0.9km of upstream habitat, totalling ~2.6km.

Figure 7.65. A pipe culvert divides the upstream and downstream sites along the Ngaruroro Backwash. **A**: The culvert's outlet is considerably smaller in size in comparison to the width of the downstream waterway. **B**: During periods of low tide, a drop forms beneath the culvert's outlet.

A pipe culvert lying beneath a dirt track road connects the Ngaruroro Backwash's upstream and downstream habitats. The culvert was measured to be approximately 9m in length with inlet and outlet diameters measuring ~0.6m (Fig.7.65A). During the site's lowest tide, a drop height of ~0.3m forms beneath the culvert's outlet in the downstream region (Fig.7.65B).

Upstream

GPS Coordinates (DD): -39.568407, 176.913389

Figure 7.66. The upstream habitat along the Ngaruroro Backwash. **A**: Long grasses, mature shrubs and sparse tree cover span the left bank's riparian buffer. **B**: Mature shrubs and a receding stream bank form a wetland habitat during periods of high tide.   **C**: Floating macrophytes collect around the right bank's shrubs. **D**: A slumping bank due to recent erosion around the culvert.

Averaging a width of 10.2m and a depth of 0.96m, the Ngaruroro Backwash's upstream sub-site has little hydraulic heterogeneity, with a slow flow of water occurring throughout the day. Few large boulders and cobbles are present onsite, resulting in the lack of cascades or riffles, suitable habitat for Ephemeroptera, Plecoptera and Trichoptera (EPT) invertebrate colonisation. Approximately 60% of the stream bed is covered by deposited fine sediment like sand or silt though scattered woody debris and small boulders line large areas of the stream bank.

The riparian buffer along the left and right banks are constrained by between 7 and 10m of mixed vegetation. Along the left bank, tree cover is sparse and scattered amongst a diversity of long grasses and mature shrubs which span the width of the buffer and run down to the water's edge (Fig.7.66A). A small amount of overhanging vegetation may provide suitable resting or spawning habitat for aquatic species. Mature shrubs, sedges and tussocks span large areas of the right bank, though tree cover remains sparse (Fig.7.66B). During periods of high tide, the stream's edge recedes back into the riparian strip, forming a wetland habitat which in some areas joins an adjacent, parallel waterway. This provides ample cover and breeding grounds for fish between root mats and beneath overhanging vegetation. Due to the lack of tree cover along this site, little shade is cast onto the waterway throughout the day, resulting in a high abundance of submerged macrophytes which span the width of the stream. Floating and submerged macrophytes provide large areas for fish to rest or hide from predators, though if volumes become too great, the waterway risks becoming too choked for fish to inhabit (Fig.7.66C). Recent bank erosion around the culvert could be due to scouring at the water line as the water level rises during high tides. Increased pressure as heavy farming vehicles pass over the dirt track road may also contribute to the slumping of the bank around the culvert (Fig.7.66D).

Downstream:

GPS Coordinates (DD): -39.568111, 176.918613

Figure 7.67. The downstream habitat along the Ngaruroro Backwash. **A**: Slow water flow across shallow areas lined with cobbles, boulders and woody debris. **B**: Thick mud banks covered in filamentous algae become exposed during a low tide. **C**: Long grasses and mature shrubs span the left and right bank's riparian buffers. **D**: Eroded banks are numerous along the downstream habitat.

With an averaging width of 25.4m, the downstream site of Ngaruroro Backwash is considerably larger than its upstream site above the culvert, though its average depth remains similar at 0.68m during a low tide. There is diverse hydraulic heterogeneity within the downstream site as a fast water flow through the culvert creates a rapid cascade beneath the outlet during low tides. Water flow remains slow around the banks however, as large cobbles, boulders, woody debris and root mats line the stream bed in shallower areas, forming riffles and deep pools with varying degrees of turbulence (Fig.7.67A). Beyond the banks, sand, gravel and thick mud covered in filamentous algae line the stream bed which become exposed during periods of low tide or little rainfall (Fig.7.67B). The diversity of substrate types within the waterway provide suitable resting and feeding habitat for migrating fish, as well as favourable breeding grounds for EPT invertebrates.

As with the upstream site, the riparian buffer along both banks span between 7 and 10m in width and are constrained by varied vegetation and farming fences. Both the left and the right banks consist mainly of mature shrubs, sedges, tussocks and long grasses with little tree cover (Fig.7.67C). Unlike the upstream site, the riparian vegetation does not run down to the water's edge, rendering the bank unprotected and vulnerable to erosion. Approximately 75% of the stream bank shows signs of recent or active erosion, potentially due to scouring at the water line and slumping of the grassy bank (Fig.7.67D). Due to the lack of overhanging vegetation and tree cover, ~85% of the stream bed

is exposed to direct sunlight throughout the day, stimulating the growth of submerged macrophytes in deeper areas and filamentous algae in shallower areas.

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 87.41) yields a Pearson-chisq dispersion value of 4.82 (p = 5.7E-07) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 70.85) yields a Pearson-chisq value of 1.55 (p = 0.12) = minimal overdispersion exists. An additional Vuong AIC z statistic of 6.16 (p = 3.71E-10) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 1008.10) yields a Pearson-chisq dispersion value of 9.78 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 423.41) yields a Pearson-chisq statistic of 0.60 (p = 1) = overdispersion does not exist. An additional Vuong AIC z statistic of -0.93 (p = 0.18) shows that the **zero-inflated negative binomial GLM** (AIC = 418.20) is favoured over a negative binomial GLM as zero-inflation does occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 109) is significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 9) along the Ngaruroro Backwash (p = 2.75E-07) (Fig.7.68). An IRR value of 12.11 for the US sub-site (95% CI: 4.68 – 31.35) is considerably higher than an IRR value of 1.50 for the DS sub-site (95% CI: 0.67 – 3.36), showing that the inanga abundance is higher within the upstream sub-site situated above the culvert, and that the culvert does not act as a barrier to upstream migrating fish.

The total abundance of all species in the US sub-site (N = 229) is also significantly higher than the total abundance of all species in the DS sub-site (N = 94) along the Ngaruroro Backwash (p = 5.87E-04) (Fig.7.68). The IRR value for the US sub-site is 3.02 times (95% CI: 1.61 – 5.67) larger than the DS sub-site IRR value of 3.16 (95% CI: 1.91 – 5.23), showing that a higher total abundance of all species inhabit the upstream reaches of the stream above the culvert, and that the structure does not restrict the upstream passage of fish. The estimated odds ratio (OR) of observing an excess zero within the US data was 1.48 times the odds of there being an excess zero in the DS data (OR = 1.02, p = 0.40).
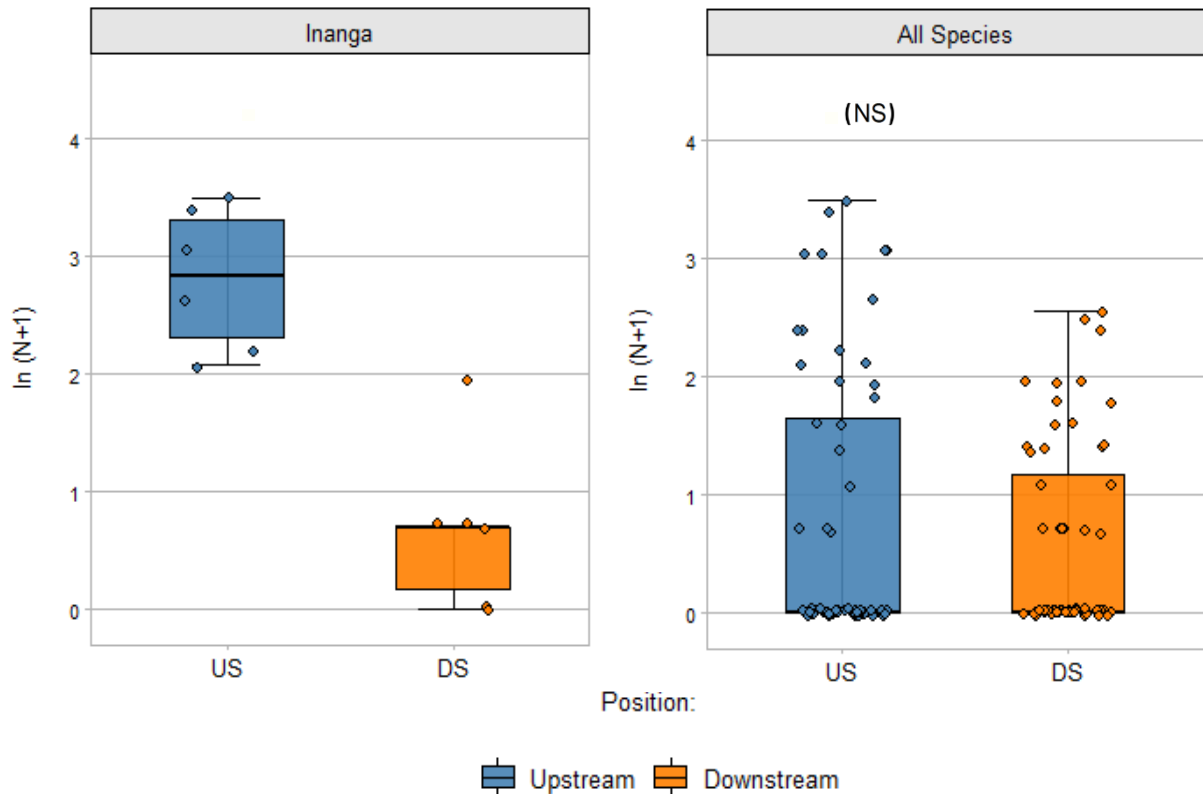
Figure 7.68. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Ngaruroro Backwash. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance level without brackets indicate count model p-values, significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: *Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.*

H_1: *Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.*

The Shannon Weiner diversity index (H') of the US sub-site (H' = 1.27) along the Ngaruroro Backwash is significantly different to the H' value of the DS sub-site (H' = 1.68) as an US ENS value of 3.56 (H'$_{max}$ = 1.79, p = 2.80E-08) and a DS ENS value of 5.35 (H'$_{max}$ = 1.79) shows that there is a greater biodiversity within the downstream reaches of the stream and that the culvert may act as a partial barrier to some smaller-bodied fish with weaker swimming abilities.

Pielou's evenness index (J') is lower in the US sub-site (J' = 0.71) compared to the evenness within the DS sub-site (J' = 0.94), although the species richness (S) was the same within both US and DS reaches (S = 6). A Sørensen's similarity index of 1.00 indicates that 100% of species present across the site were found within both sub-sites. The migratory inanga and yellow-eye mullet, the resident common bully, crab and New Zealand freshwater shrimp, and the invasive mosquitofish are the six species present within both of the US and the DS sub-sites.

164

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, dissolved oxygen and specific conductivity measures are similar within both sub-sites, suggesting that the pipe culvert drainage structure does not act as an environmental barrier and that the abundance of inanga and the total abundance of all species has not been affected by any fluctuations in environmental factors along the Ngaruroro Backwash.

7.2.08. Tutaekuri Waimate Stream (twst)

<u>Site Description</u>

<u>Date of Study:</u>

Wednesday 4th December 2019.

<u>Weather Conditions:</u>

High sunshine levels with little cloud cover; high air temperature throughout the day with no breeze.

<u>Hydrographic Tide Predictions:</u>

Non-tidal site. Lowest tide of 0.4m at 06:09 rising to 1.6m by 12:26 (LINZ, 2020).
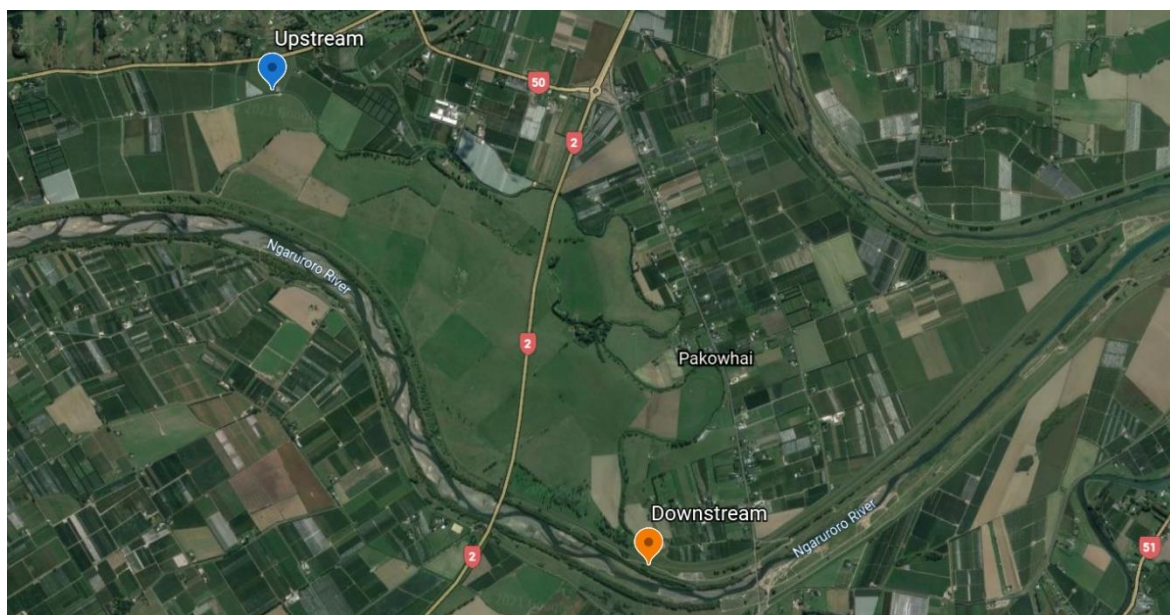
Figure 7.69. Google Earth image showing the location of the studied upstream and downstream Tutaekuri Waimate Stream sub-sites in relation to the Ngaruroro River, State Highways 2 and 50 and the village of Pakowhai (Google Earth, 2021).

Branching from the Ngaruroro River, the Tutaekuri Waimate Stream meanders in a northerly direction through the rural village of Pakowhai in Central Hawke's Bay, before continuing west and becoming the Paherumanihi Stream (Fig.7.69). Running parallel to State Highway 2, the downstream sub-site lies 0.46km from the Ngaruroro River and 7.3km from the open ocean and is surrounded on both sides by agricultural land. The upstream sub-site is located within viticultural land and runs parallel to State Highway 50. There is >18km of suitable and unrestricted habitat directly above the upstream sub-site with an additional >12.6km of available habitat from six tributaries of the Tutaekuri Waimate Stream. Therefore, there is at least 30.6km of potential inanga habitat upstream of the studied drainage structure.

Permissions were required to access and set nets within the privately-owned upstream sub-site. On the day of study, it was also required that signatures be taken to sign in and out of the site for health and safety purposes. Access to the downstream sub-site required the use of the HBRC master key to open a locked gate. Security was sufficient along both sub-sites as neither were accessible to the general public.

Drainage structure present:

Figure 7.70. Two culverts with attached flood gates divide the upstream and downstream sub-sites along the Tutaekuri Waimate

Two pipe culverts, each spanning ~37m in length, run beneath a dirt track road, dividing the upstream and the downstream sub-sites along the Tutaekuri Waimate Stream. A flood gate is attached to the outlet of each of the culverts, totalling 5.5m in width (Fig.7.70).

Upstream

GPS Coordinates (DD): -39.566647, 176.821831
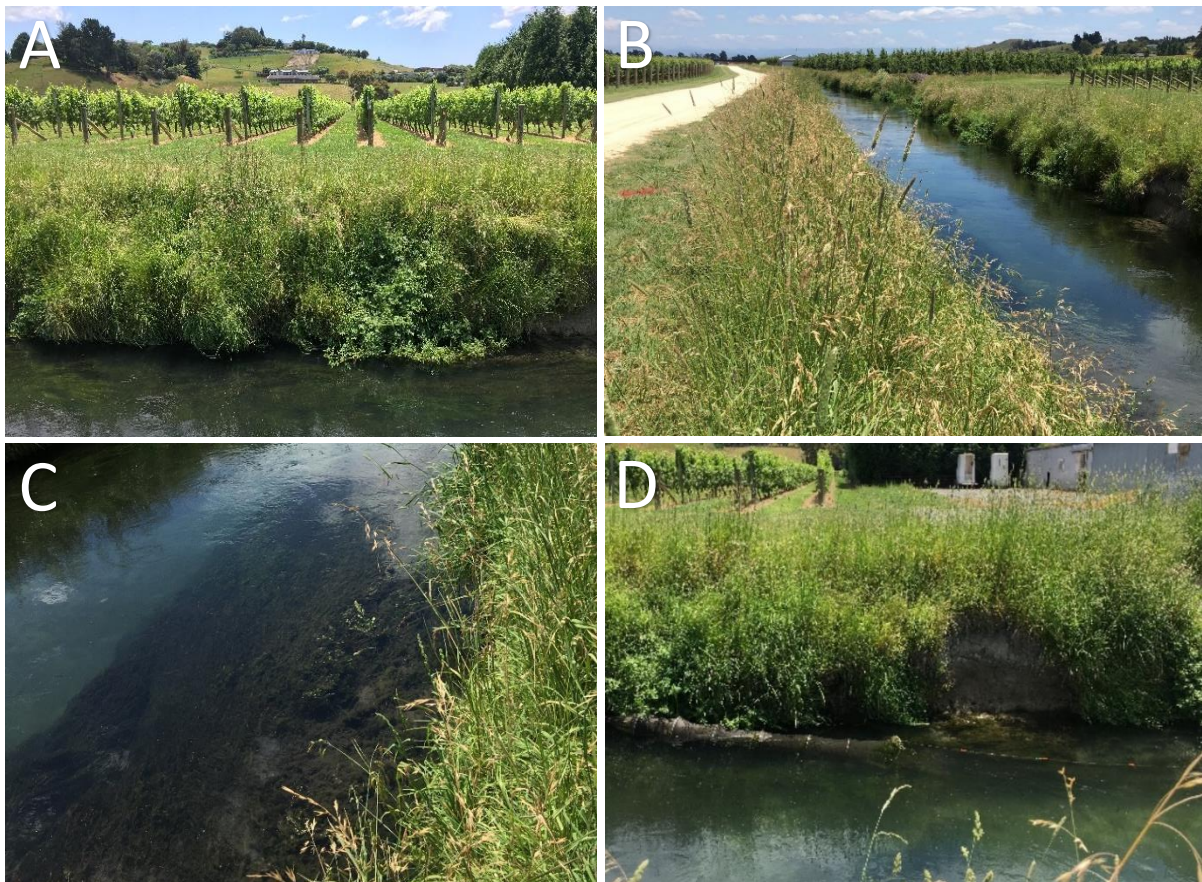
Figure 7.71. The upstream habitat along the Tutaekuri Waimate Stream. **A**: Shrubs and long grasses along the left bank encroach into the stream. **B**: A dense strip of long grasses span the width of the right bank. **C**: Submerged macrophytes span ~60% of the stream. **D**: Erosion has led to undercut banks and landslips in some regions.

The upstream sub-site along the Tutaekuri Waimate Stream is mainly a backwater with a number of lentic pools of varying depths and little turbulence. A dense riparian strip of shrubs and long grasses span the 3-4m wide left bank which encroaches into the stream and provides an abundance of suitable fish spawning habitat (Fig.7.71A). Young exotic, long grasses span the ~4m width of the right bank which also encroaches into the stream whilst a dirt track road and a strip of heavily mown grass lies above the dense riparian vegetation (Fig.7.71B). Shading is cast across ~25% of the stream along each bank, though large areas of the sub-site remain exposed to direct sunlight throughout the day. Subsequently, this has led to the excessive growth of submerged macrophytes which span ~60% of the stream's length, providing copious amounts of habitat for fish to shelter, feed or seek refuge from predators (Fig.7.71C).

Although each bank's overhanging riparian vegetation offers a small level of protection against hydrodynamic scouring, approximately 50% of each bank shows signs of active or recent erosion. Fluctuating water levels during periods of high rainfall has led to the undercutting of banks and therefore a number of significant landslips have occurred along the left bank (Fig.7.71D). Large quantities of small boulders, cobbles and plant matter have fallen into the stream beneath the submerged macrophytes, covering ~60% of the stream bed. The fallen substrates have remained clear of filamentous algae and have provided favourable conditions for EPT invertebrate colonisation.

Downstream

GPS Coordinates (DD): -39.597135, 176.863051
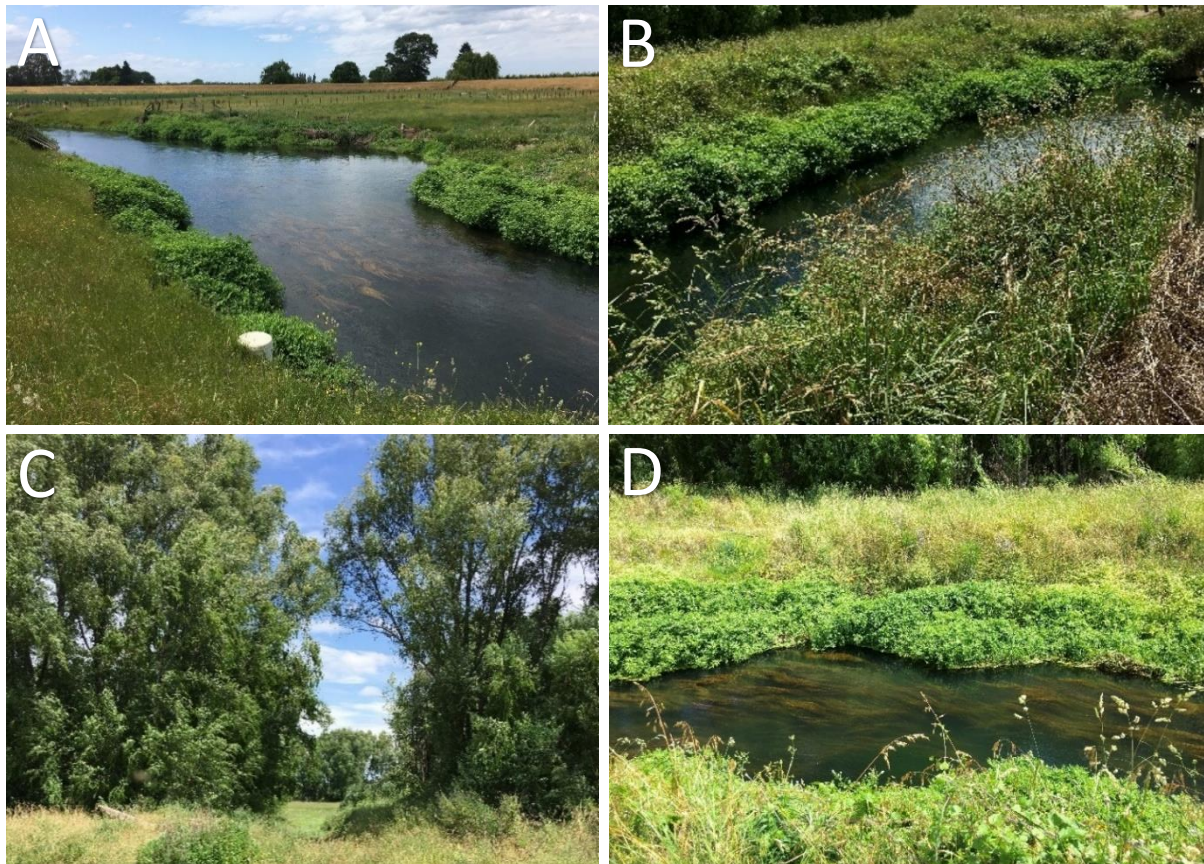
Figure 7.72. The downstream habitat along the Tutaekuri Waimate Stream. **A**: Long grasses and aquatic plants line the upper reaches of the left and the right banks. **B**: Shrubs and long grasses encroach into the lower reaches of the stream. **C**: Mature, native trees lie behind the left bank. **D**: Submerged macrophytes choke at least 95% of the water channel.

The downstream sub-site along the Tutaekuri Waimate Stream has a number of pools of varying depths and a strong water flow with some significant turbulence across the sub-site, despite the settled climatic conditions that were experienced on the day of the study. A dense strip of long grass spans each of the ~5m wide left and right banks along the upper reaches of the sub-site. Mature shrubs line the water's edge along each bank and encroach into the stream, which provides an abundance of suitable spawning habitat for migrating or resident fish (Fig.7.72A). The riparian vegetation along the lower reaches of each bank differs slightly to the upper; diverse and natural shrubs, aquatic plants and long grasses span the 7-10m wide lower left bank, which encroaches into the stream (Fig.7.72B). Mature, native trees lie behind the left bank and cast some shading across the stream at certain periods of the day (Fig.7.72C). A dense riparian strip of diverse long grasses span the ~5m wide lower right bank, overhanging in some areas and offering a high volume of additional fish spawning habitat. Approximately 25% of each bank shows signs of active erosion as the daily fluctuations of the tide have caused scouring at the water line in some areas, though the dense encroaching bank vegetation acts as a defence against any significant bank undercutting and subsequent landslips.

Shading is cast across only 10% of the stream from each bank, leaving much of the sub-site exposed to direct sunlight at all times. This has subsequently resulted in the excessive growth of submerged macrophytes which choke >95% of the water channel, providing an abundance of suitable habitat for aquatic species to take shelter and rest (Fig.7.72D). Beneath the dense macrophyte cover, fine substrates like gravel, sand and cobbles line ~60% of the stream bed, though very little favourable habitat for EPT invertebrate colonisation is available.

Results

Note: Due to the high level of turbulence, the fast water flow and the large water depths of the downstream sub-site, it was deemed unsafe for fyke netting to be set within the stream. Therefore, no downstream data was collected, statistical comparisons could not be made and the passability of the flood gate could not be reliably assessed.

7.2.09. Pakowhai (paps)

Site Description

<u>Date of Study:</u>

Friday 6th December 2019.

<u>Weather Conditions:</u>

Mostly clear skies with high levels of sunshine; warm air temperature increasing throughout the day.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.4m at 07:46 rising to 1.6m by 14:01 (LINZ, 2020).
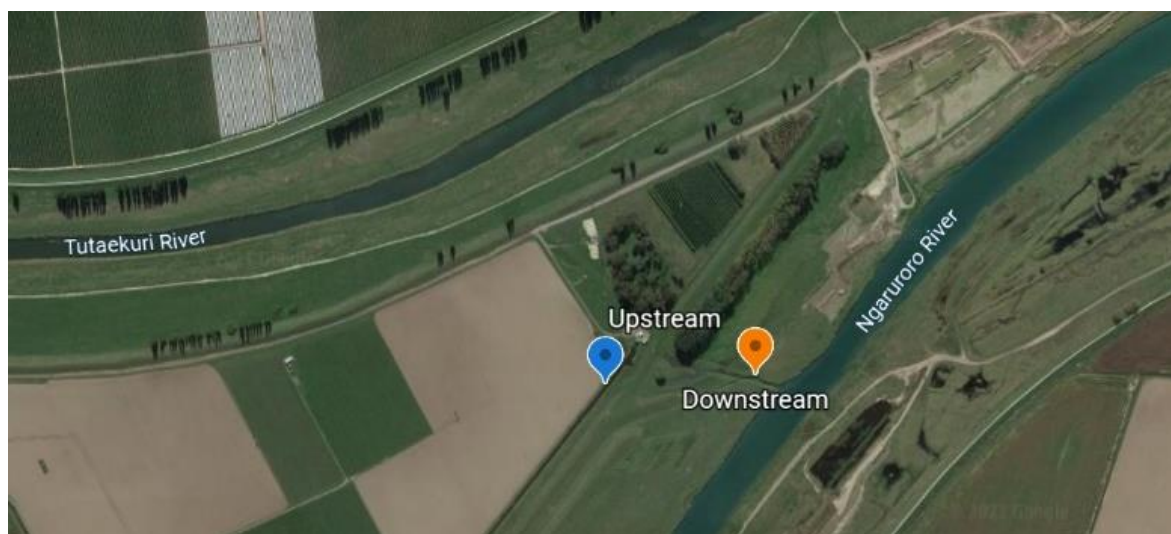
Figure 7.73. Google Earth image showing the location of the studied upstream and downstream Pakowhai sites in relation to the Ngaruroro and the Tutaekuri Rivers (Google Earth, 2021).

Branching west from the Ngaruroro River, the Pakowhai sites lie south of the Tutaekuri River and east of the suburban village of Pakowhai in central Hawke's Bay (Fig.7.73). Agricultural land surrounds the downstream sub-site, which lies ~20m from the main Ngaruroro channel and 2.7km from the open ocean. The upstream sub-site runs parallel to active viticultural land and continues in a south westerly direction before meandering north through Pakowhai village for a total of 5.4km. A small waterway branches from the Pakowhai tributary, running in a south westerly direction for an additional 2.4km, totalling 7.8km of available upstream habitat.

Access to the sub-sites proved challenging as dirt track roadways did not continue down to the stream's edge, therefore it was required that nets were carried by hand for some distance through overgrown agricultural land. Permissions were not required to access the site as locked gates were opened with the use of the HBRC master key.

Drainage structure present:

Figure 7.74. A flood gate and pipe culvert lie between the upstream and downstream sites along Pakowhai. **A**: The closed flood gate during a low tide. **B**: The inlet of the large pipe culvert, spanning ~2m in diameter.

A flood gate and a pipe culvert lie between the upstream and the downstream Pakowhai sub-sites. The flood gate is attached to the outlet of the large culvert (Fig.7.74A) which lies beneath a steep grassy track. The flood gate has a height of ~2.5m and is fixed across the ~2m culvert outlet. The culvert itself spans 32m in length and connects the upstream and downstream habitats when the flood gate opens (Fig.7.74B). A pumping station measuring ~8m in length is present onsite, though this structure divides the upstream sub-site from a separate downstream sub-site which will not be assessed for inanga abundance, therefore the pumping station will not be assessed in this study.

Upstream

GPS Coordinates (DD): -39.572025, 176.898689
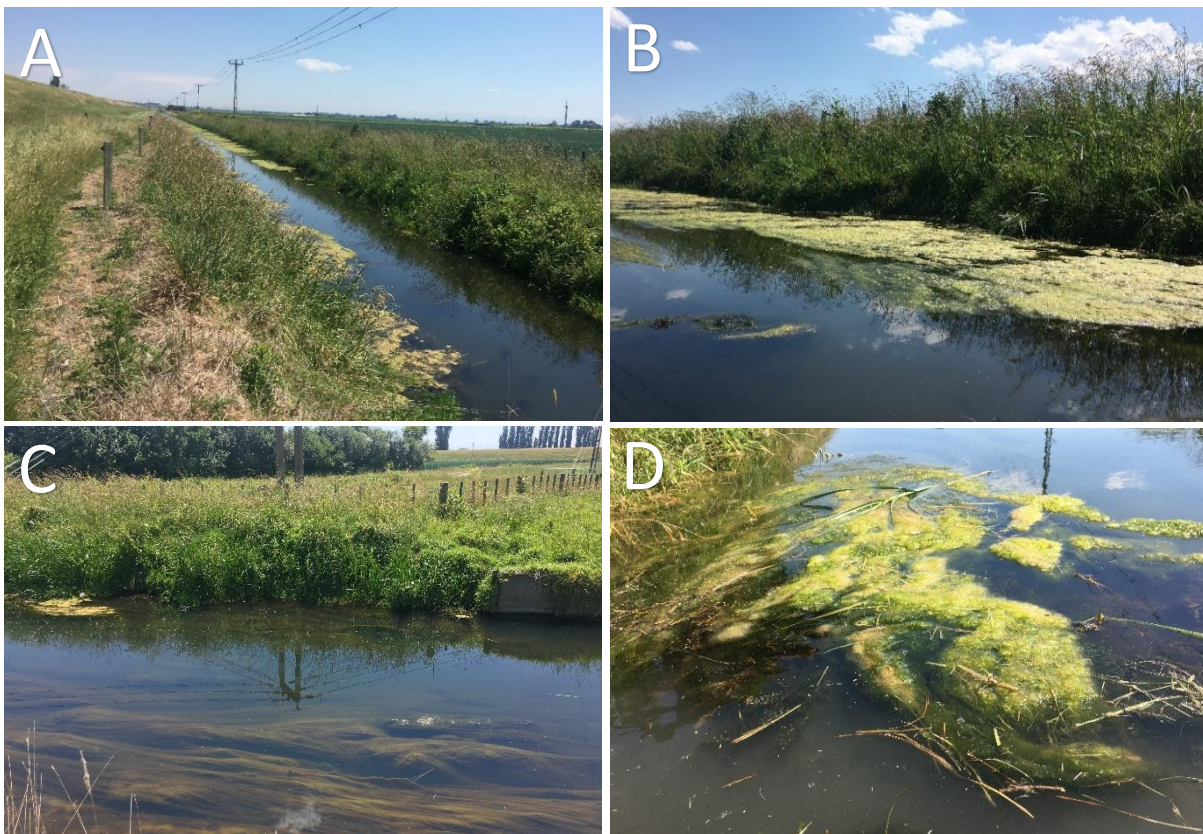
The upstream Pakowhai sub-site has very little hydraulic heterogeneity as diverse vegetation and substrate types are sparse. Constrained by barbed wire fences, the left and the right banks span ~7m and ~4m in width, respectively. Along the left bank, sedges and tussocks encroach the water's edge and are scattered amongst patches of heavily mown grass (Fig.7.75A). Mature shrubs and young, exotic grasses span the right bank, casting shade along ~10% of the stream (Fig.7.75B). As tall trees are not present along either bank, most of the sub-site is exposed to sunlight at all times throughout the day, providing suitable conditions for excessive macrophyte growth. Approximately 75% of the water channel is choked with dense submerged macrophytes spanning the depth of the stream (Fig.7.75C). Additional floating macrophytes have become entangled in encroaching riparian vegetation and have collected around the stream's banks (Fig.7.75D). Although submerged macrophytes can act as suitable resting sites for some fish species, they have considerably slowed down the sub-site's water flow and have reduced its hydraulic complexity. Riffles, cascades and pools are not present onsite as the stream bed is covered by fine sediment, rather than varied substrate types like boulders and cobbles, reducing the amount of suitable habitat for EPT colonisation.

Downstream

GPS Coordinates (DD): -39.572212, 176.901408
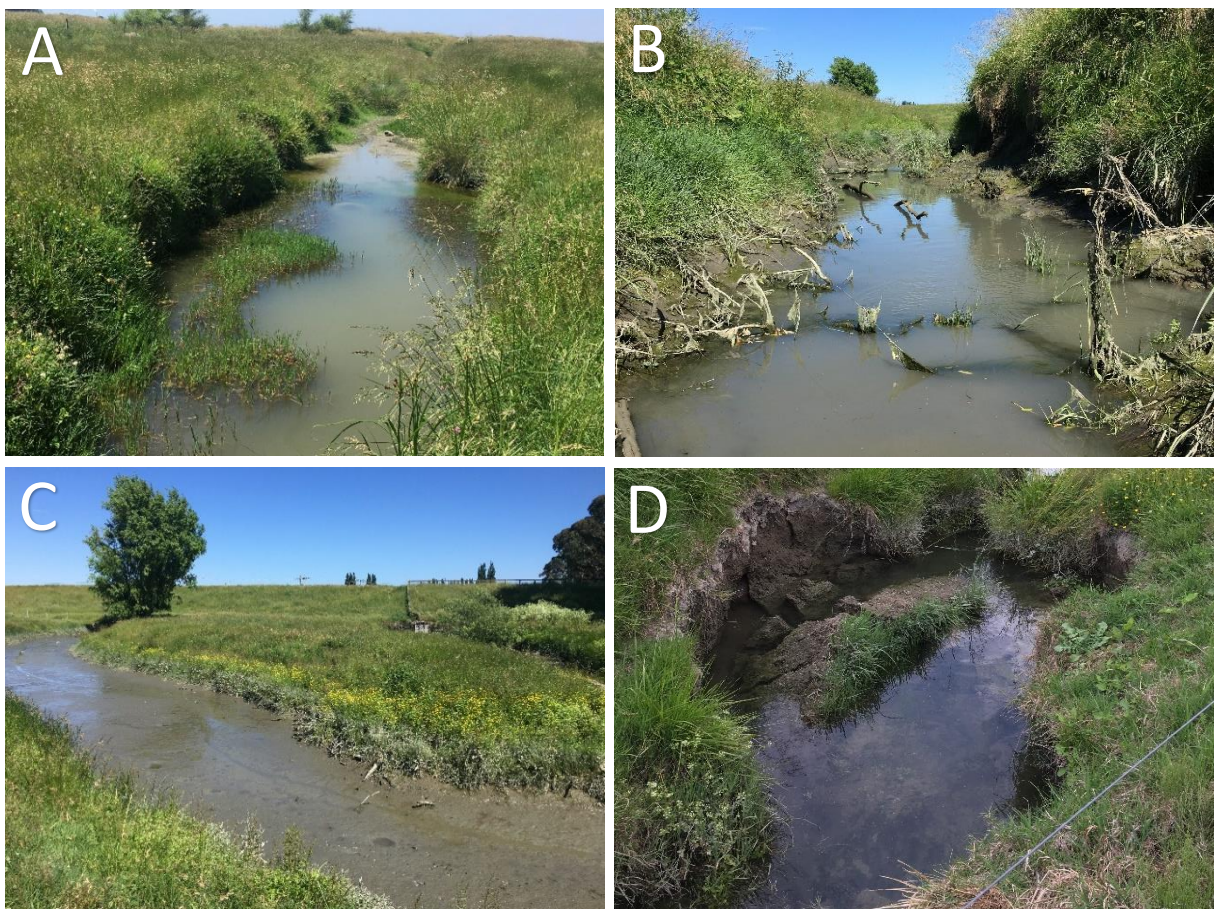
Figure 7.76. The downstream habitat along the Pakowhai site. **A**: Mature shrubs and long grasses span both the left and the right banks. **B**: Woody debris, root mats and plant matter are abundant along the lower reaches of the sub-site. **C**: The stream is reduced to a small rill during a low tide. **D**: Semi-isolated rockpools branch from the sub-site.

The downstream Pakowhai sub-site has little hydraulic heterogeneity, though the hydraulic components onsite differ considerably to those along the upstream sub-site. The left and the right banks both span ~7m in width and are constrained on each side by loose barbed wire farming fences. Tree cover is sparse and scattered amongst mature shrubs and young, exotic grasses which span the width of each bank (Fig.7.76A). The riparian vegetation along the upper reaches of the sub-site encroaches into the water, though undercut banks and root mats are abundant along the lower reaches. Woody debris and plant matter, fallen from the banks, line the stream bed and provide suitable habitat for fish and invertebrate species to cover and seek refuge (Fig.7.76B). Approximately 70% of the waterway is scattered with boulders, cobbles and gravel, substrates which are suitable for EPT colonisation. Due to the lack of tree cover, little shade is cast across the stream, though macrophyte growth remains sparse.

The sub-site experiences a considerable change in water level throughout the day, though water flow remains slow at all times. During the lowest tide, the upper reaches dry almost completely, and the wide stream is reduced to a small rill (Fig.7.76C). The water level along the lower reaches reduces also, exposing the woody debris and root mats which lie beneath the water's surface. As the tide changes, the water levels increase across the sub-site, submerging all of the plant matter once again. The daily fluctuations of the tide causes erosion of the stream bank due to scouring at the water line. Around 65% of each bank shows active or recent erosion, with some areas having significant landslips. Branching from the lower reaches of the sub-site are multiple semi-isolated rockpools which retain a deeper water level than the main stream channel, though the hydraulic conditions in these areas are similar to those across the sub-site (Fig.7.76D).

Results

Note: Only four fyke nets were set along the downstream sub-site as water levels were too shallow in some areas for efficient net setting.

Inanga and All Species Total Abundance

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

Model Justification:

Inanga abundance analysis:

A suitable, well-fitting Poisson GLM (AIC = 4) yields a Pearson-chisq dispersion value of 3.49E-11 (p = 1) = overdispersion does not exist, showing that the **Poisson GLM** is the favoured model to use.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 373.87) yields a Pearson-chisq dispersion value of 9.90 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 152.69) yields a Pearson-chisq statistic of 0.86 (p = 0.84) = overdispersion does not exist. An additional Vuong AIC z statistic of 5.84 (p = 2.64E-09) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 0) along the Pakowhai stream (p = 1) (Fig.7.77). The IRR value for the US sub-site is 1 times (95% CI: 0 – infinite) the IRR value of 0.09 (95% CI: 0 – infinite) for the DS sub-site, showing that inanga were not found to inhabit either sub-site along the stream. Wide US and DS CIs arose due to the small sample sizes of each of the two sub-sites.

The total abundance of all species in the US sub-site (N = 23) is not significantly higher than the total abundance of all species in the DS sub-site (N = 46) along the Pakowhai stream (p = 0.19) (Fig.7.77). An IRR value of 0.33 for the US sub-site (95% CI: 0.06 – 1.72) is not higher than an IRR value of 1.15 for the DS sub-site (95% CI: 0.33 – 3.99), showing that a higher total abundance of all species inhabit the downstream reaches of the stream and that the upstream passage of fish is restricted by the flood gate drainage structure.
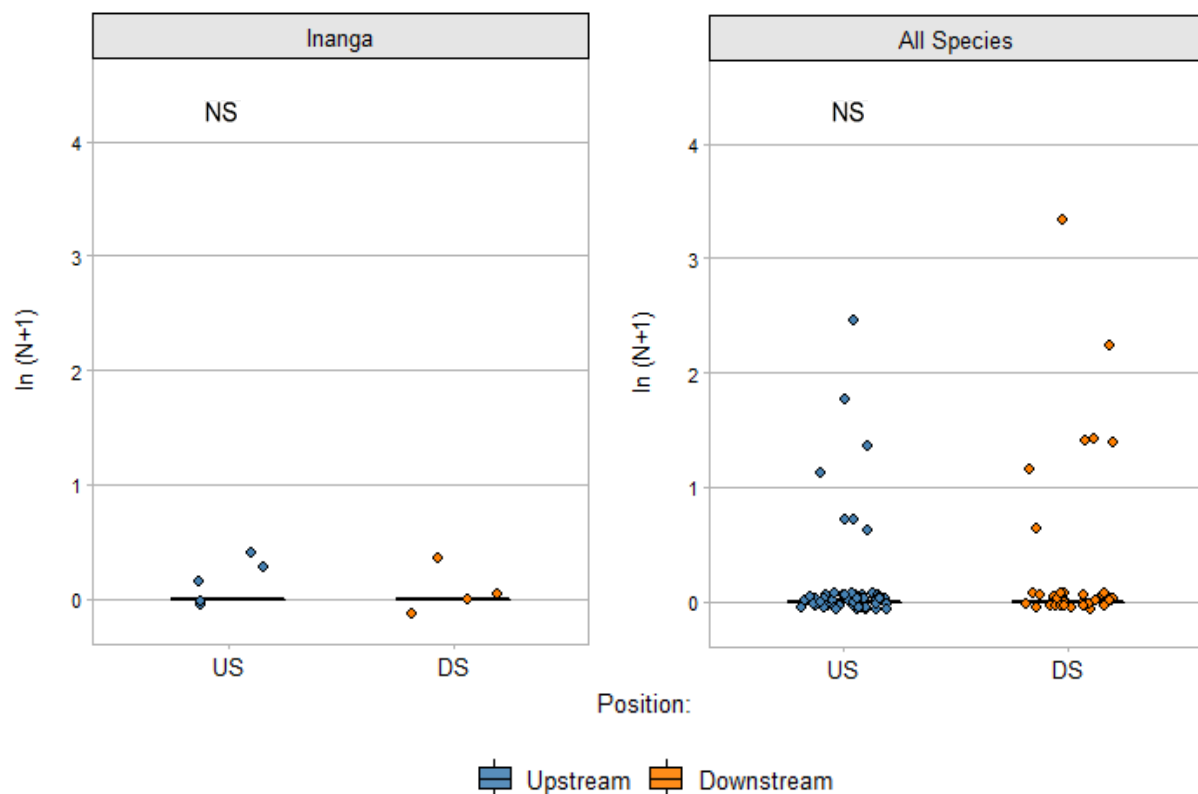
Figure 7.77. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Pakowhai site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (Poisson GLM for 'Inanga' and negative binomial GLM for 'All Species').

Species Diversity and Similarity

H$_0$: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H₁: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.36) along the Pakowhai site is significantly different to the H' value of the DS sub-site (H' = 0.82) as an US ENS value of 1.43 (H'_{max} = 1.10, p = 0.03) and a DS ENS value of 2.26 (H'_{max} = 1.10) shows that there is a greater biodiversity of species within the downstream sub-site and that upstream fish passage is restricted by the flood gate barrier.

Pielou's evenness index (J') is lower in the US sub-site (J' = 0.32) than the evenness in the DS sub-site (J' = 0.74), although the US species richness (S) (S = 3) is the same as the richness DS (S = 3). A Sørensen's similarity index of 0.33 indicates that only 33% of species present across the site are found within both sub-sites. The invasive mosquitofish is the only species present within both of the US and the DS sub-sites, whereas migratory species like yellow-eye mullet and the resident common bully are present within the DS sub-site only.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature measures are similar within both sub-sites, although the dissolved oxygen saturation and the specific conductivity is higher downstream. As the flood gate acts as a barrier between the two sub-sites, the upstream flow is slowed considerably, preventing the tidal mixing of nutrients across the site and reducing the DO saturation. The highly tidal downstream sub-site and the close geographical proximity to the turbulent main channel of the Ngaruroro River has led to increased nutrient mixing downstream, raising DO and SPC levels and providing suitable conditions and habitat types for inanga to inhabit before continuing their upstream migrations, assuming that populations of inanga were present along the stream and that remediations of the flood gate barrier allowed the full passage of upstream migrating fish.

7.2.10. Memorial Park Drain (mpdr)

Site Description

Date of Study:

Tuesday 7th January 2020.

Weather Conditions:

High sunshine levels with patchy cloud cover throughout the day; warm air temperature with an occasional cool ocean breeze.

Hydrographic Tide Predictions:

Tidal site. Lowest tide of 0.5m at 09:33 rising to 1.5m by 15:40 (LINZ, 2020).
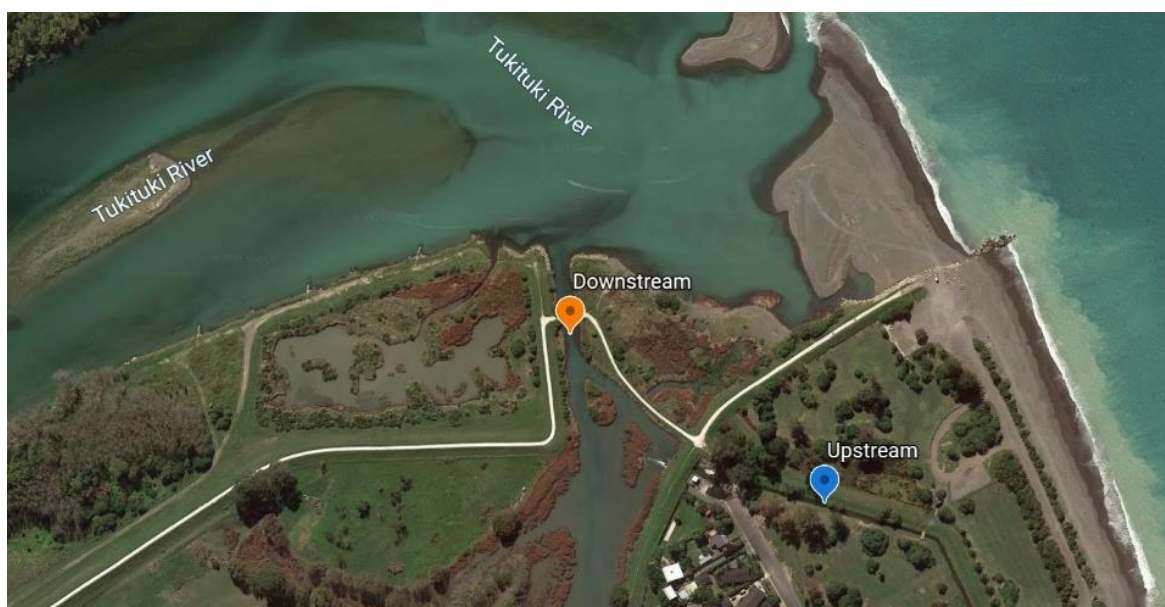
Figure 7.78. Google Earth image showing the location of the studied upstream and downstream Memorial Park Drain sub-sites in relation to the Tukituki River and the open ocean (Google Earth, 2021).

Branching south from the Tukituki River, the Memorial Park Drain lies north of the coastal village of Haumoana in eastern Hawke's Bay (Fig.7.78). The downstream sub-site runs through a public coastal cycle path and lies <50m from the main Tukituki River channel and ~0.3km from the open ocean. The waterway continues upstream in a south easterly direction for approximately 1.9km, through the Springfield Road Reserve and running parallel to Grange Road North. The stream flows through two large natural ponds, spanning 200mx35m and 350mx80m, providing additional, unobstructed available upstream habitat.

Both sub-sites ran through public areas, therefore access did not require a permit or landowner permissions. The HBRC master key was required to gain vehicle access to the downstream sub-site. Caution was essential whilst driving along the public cycle path when pedestrians or cyclists were within close proximity to the vehicle. Regular checking of the nets throughout the day ensured that the sites' security remained uncompromised.

Drainage structure present:

Figure 7.79. A flood gate and a pipe culvert divide the upstream and downstream sub-sites along Memorial Park Drain. **A**: The flood gate covers the width of the culvert's outlet. **B**: The culvert's inlet allows upstream water to drain into the downstream sub-site when the flood gate opens.

A flood gate and a pipe culvert lie between the upstream and the downstream sub-sites along the Memorial Park Drain. The flood gate has a height of ~2.5m and is attached onto the outlet of the 1.5m width culvert, beneath a steep grassy pathway (Fig.7.79A). The culvert has a length of 13m and connects the upstream and downstream sub-sites when the flood gate opens (Fig.7.79B). A small pumping station is also present onsite, though during the summer months when rainfall levels are at their lowest, the flood gate acts the site's primary flood defence mechanism. At the time of data collection, the pumping station was undergoing essential maintenance work, therefore the flood gate was the only structure assessed in this study.

Upstream

GPS Coordinates (DD): -39.60118, 176.946123
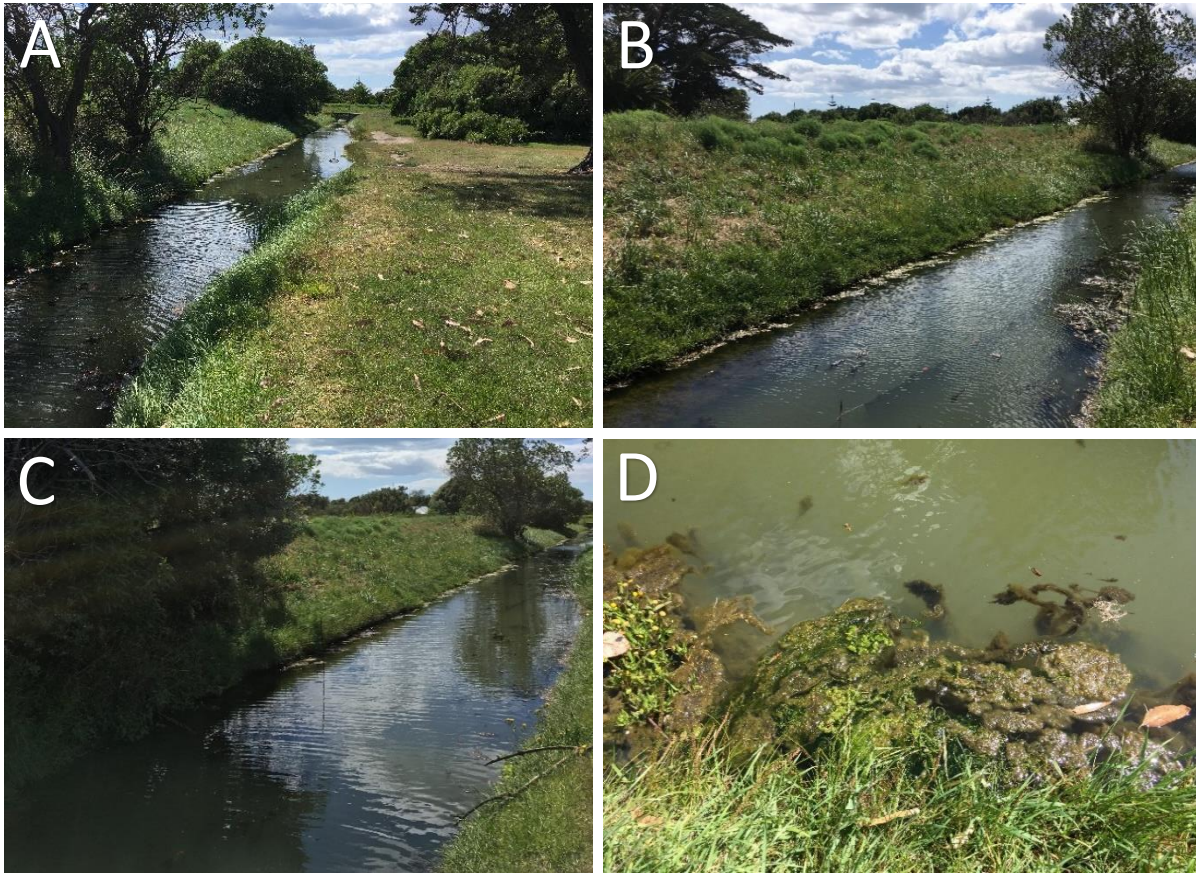
Figure 7.80. The upstream habitat along Memorial Park Drain. **A**: Heavily mown grass spans the width of the left bank. **B**: Mature native trees are scattered amongst heavily mown grass along the right bank. **C**: Large overhanging trees cast shading across the width of the stream and have led to increased bankside erosion. **D**: Macrophyte 'shelves' have formed along each bank.

The upstream sub-site along the Memorial Park Drain has little hydraulic heterogeneity and the diversity amongst the riparian vegetation is minimal. Spanning <1m in width, heavily mown grass lines the left bank, casting little shading across the stream (Fig.7.80A). A thin strip of long grass encroaches into the waterway from the left bank, providing a small amount of suitable fish cover or resting habitat. The right bank has a width and a height of ~3m from the water's surface. Mature native trees are sparsely scattered amongst heavily mown grass (Fig.7.80B), casting significant shading across the width of the stream. The slumping of the overhanging trees has led to the active erosion of ~25% of the right bank (Fig.7.80C), though scouring at the water line has eroded regions of both banks where encroaching vegetation is least abundant and the banks are the most exposed.

The water flow downstream has been considerably slowed due to the high abundance of submerged macrophytes which choke the waterway. Excess submerged and floating macrophytes have accumulated along each bank forming 'shelves' and trapping additional fallen plant matter and woody debris (Fig.7.80D). The stream's undercut banks and high level of macrophytes provide suitable refuge habitat for small bodied fish species, though the lack of diverse hydraulic components like riffles, cascades and waterfalls limits the amount of favourable habitat for EPT colonisation.

GPS Coordinates (DD): -39.600231, 176.943871

Figure 7.81. The downstream habitat along Memorial Park Drain. **A**: Canopy trees are scattered amongst dense grasses along the left bank. **B**: Long grasses act as a defence against significant bank erosion. **C**: Mature shrubs and sedges are scattered amongst gravel and patches of mown grass. **D**: Large boulders become a rockpool habitat during a high tide.

Unlike the upstream sub-site, the downstream sub-site along the Memorial Park Drain has some hydraulic heterogeneity and there is significant diversity amongst substrate types and riparian vegetation. Dense long grasses and mature shrubs line the 10m wide left bank, spanning down to the water's edge. Canopy trees are set within the centre of the riparian strip and are scattered amongst the dense vegetation, though very little shading is cast across the stream (Fig.7.81A). Fluctuating water levels along the tidal site have caused scouring at the water line, though long grasses along the left bank have acted as a defence against significant erosion (Fig.7.81B). A diverse range of mature shrubs and sedges span the 10-15m width of the right bank and are scattered amongst gravel and patches of mown grass (Fig.7.81C). During a high tide, a small amount of shading is cast across the stream, though when the tide recedes, shading is mainly cast across the exposed regions of the stream bed. Approximately 30% of the right bank shows signs of recent erosion as the fluctuating water levels scour the less protected bank.

Due to the lack of significant shading across the stream, macrophytes span ~40% of the waterway, providing suitable resting habitat for migrating fish. A fast flow with moderate turbulence remains along the lower reaches of the sub-site which, during a low tide, forms riffles and cascades over small sized rocks and cobbles; hydraulic components which are favourable for EPT colonisation. Large boulders have been artificially placed in a cluster along the left bank, which forms a rockpool microhabitat when partially submerged during a high tide (Fig.7.81D). These areas with a slower

water flow and less turbulence provide additional resting, feeding and spawning habitat for migrating fish.

Results

<u>Inanga and All Species Total Abundance</u>

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

<u>Model Justification:</u>

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 91.13) yields a Pearson-chisq dispersion value of 10.7 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 31.67) yields a Pearson-chisq value of 0.45 (p = 0.92) = overdispersion does not exist. An additional Vuong AIC z statistic of 1.69E+4 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 4987.08) yields a Pearson-chisq dispersion value of 86.99 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 327.54) yields a Pearson-chisq statistic of 0.42 (p = 1) = overdispersion does not exist. An additional Vuong AIC z statistic of -1.30 (p = 0.10) shows that the **zero-inflated negative binomial GLM** (AIC = 320.63) is favoured over a negative binomial GLM as zero-inflation occurs.

<u>Results:</u>

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 28) along the Memorial Park Drain (p = 1) (Fig.7.82). An IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 4.67 for the DS sub-site (95% CI: 0.76 – 28.54), showing that inanga abundances are higher downstream and that the flood gate restricts upstream inanga migrations. Wide US and DS CIs arose due to the small sample sizes of each sub-site.

The total abundance of all species in the US sub-site (N = 825) is significantly higher than the total abundance of all species in the DS sub-site (N = 249) along the Memorial Park Drain (p = 0.01) (Fig.7.82). An IRR value of 10.67 for the US sub-site (95% CI: 2.05 – 55.68) is higher than an IRR value of 8.14 for the DS sub-site (95% CI: 2.19 – 30.26), showing that a higher total abundance of all species inhabit the upstream reaches of the stream. However, additional biodiversity analyses of each sub-site are required to assess whether the higher upstream abundance comprises of native, migrating species which were able to navigate beyond the flood gate, or populations of invasive, resident species inhabiting the upstream sub-site only.

The estimated odds of obtaining an excess zero in the US data is 5.53 times the odds ratio (OR) of there being an excess zero in the DS data (OR = 0.96, p = 0.97).
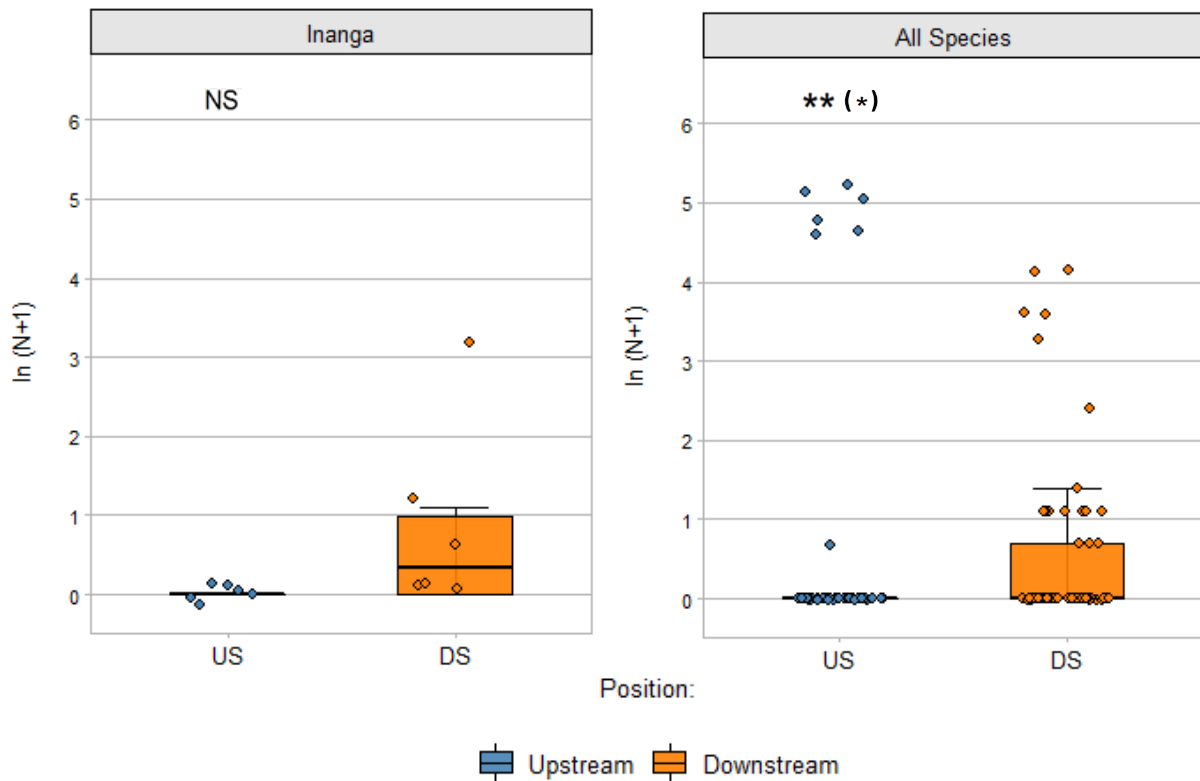
Figure 7.82. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Memorial Park Drain site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance level without brackets indicate count model p-values, significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: *Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.*

H_1: *Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.*

The Shannon Weiner diversity index (H') of the US sub-site (H' = 9.35E-03) along the Memorial Park Drain is significantly different to the H' value of the DS sub-site (H' = 0.68) as an US ENS value of 1.01 (H'$_{max}$ = 0.69, p = 8.1E-19) and a DS ENS value of 1.96 (H'$_{max}$ = 1.79) confirms that there is a considerably greater biodiversity within the downstream sub-site than the upstream sub-site and that upstream fish passage is restricted by the flood gate barrier.

Pielou's evenness index (J') and species richness (S) is lower in the US sub-site (J' = 0.01, S = 2) compared with the evenness and richness within the DS sub-site (J' = 0.38, S = 6), whilst a Sørensen's similarity index of 0.50 indicates that 50% of species present across the site are found within both sub-sites. The invasive mosquitofish and the resident New Zealand freshwater shrimp are the only two species present upstream, whereas native, migratory species like inanga and grey mullet are present within the DS sub-site only, explaining that although the total abundance of all species is higher upstream, those populations comprised only of environmentally tolerant, pest species.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the specific conductivity measures are similar within both sub-sites, although the temperature and the dissolved oxygen saturation was higher downstream, suggesting that the flood gate does also act as an environmental barrier. The highly tidal downstream sub-site and the close geographical proximity to the open ocean has led to increased nutrient mixing downstream, leading to higher DO levels than the slow flowing upstream channel situated above the flood gate. Shallower pools and riffles during a low tide across the downstream sub-site has led to increased water temperatures and provides suitable conditions for a range of estuarine fish species to inhabit and thrive in abundance.

7.2.11. Muddy Creek (mucr)

<u>Site Description</u>

<u>Date of Study:</u>

Thursday 9th January 2020.

<u>Weather Conditions:</u>

High sunshine levels and patchy cloud cover throughout the day; warm air temperature with a cool ocean breeze.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.4m at 11:12 rising to 1.6m by 17:18 (LINZ, 2020).
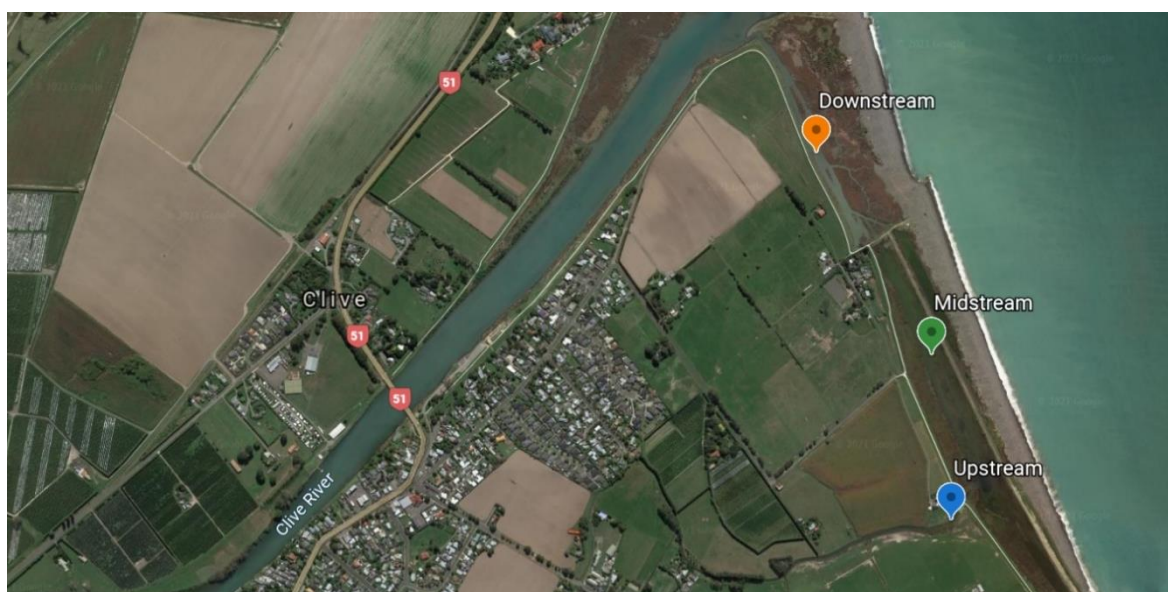
Figure 7.83. Google Earth image showing the location of the studied upstream, midstream and downstream Muddy Creek sub-sites in relation to the Clive River, the village of Clive and the open ocean (Google Earth, 2021).

Branching south from the Clive River and running parallel to the coast, the Muddy Creek site lies east of the sub-urban village of Clive and State Highway 51 (Fig.7.83). The downstream sub-site lies 0.9km from the open ocean and runs between marshland and active agricultural land. The midstream sub-site continues in a southerly direction for an additional 1.4km and has a width of ~40-65m. The upstream sub-site runs perpendicular to the coast and continues in a westerly direction through the rural villages of Whakatu and Mangateretere for ~4.3km, before separating into two smaller streams. Both distributaries continue in a south westerly direction for 2.2km and 3km, totalling approximately 9.5km of available upstream habitat.

The three sub-sites run through a rural public cycle path, therefore access did not require permits or landowner permissions, although the HBRC master key was required to gain vehicle access to all sub-sites. Regular checking of the nets throughout the day ensured that the sites' security remained sufficient in the publicly accessible site.

Drainage structure present:

Figure 7.84. Two drainage structures lie along Muddy Creek. **A**: A Fish-Friendly Flood Gate (FFFG) divides the midstream and downstream sites. **B**: A pumping station divides the upstream and midstream sites.

Two drainage structures divide the three sub-sites along Muddy Creek. A 'fish-friendly' mechanism has been installed onto a flood gate which lies between the midstream and downstream sub-sites (Fig.7.84A). Spanning 2.5m in width, the 'fish-friendly' flood gate (FFFG) acts to protect the midstream sub-site against flooding, whilst also allowing continued upstream passage for migrating fish. A pumping station measuring 9m in width divides the upstream and the midstream sub-sites (Fig.7.84B). The sole aim of this structure is to act as a flood defence as no additional fish passage mechanisms have been installed.

Upstream

GPS Coordinates (DD): -39.58454, 176.932466

The upstream sub-site along Muddy Creek runs through agricultural land and has little hydraulic heterogeneity, although there is some diversity amongst its riparian vegetation. Each spanning ~5m in width, the left and the right banks are constrained by temporary electric fences as the adjacent farmland is commonly used by livestock. Long exotic grass spans the left bank, running down to the water's edge and encroaching into the stream, providing suitable resting and spawning habitat for migrating fish (Fig.7.85A). A dense strip of mature shrubs and sedges line the right bank, reducing the amount of scouring at the water line and subsequent erosion (Fig.7.85B). Long grasses span the remainder of the right bank, although stock pugging and trampling by livestock over some areas of the riparian vegetation were evident. Riparian shade was cast across ~15% of the stream (Fig.7.85C), though macrophyte growth, and therefore fish cover remained minimal.

Approximately 50% of the stream bed is covered by fine sediment, like sand and silt, and large substrates, like boulders and cobbles are sparse. Riffles, cascades and waterfalls have been replaced by a slow flowing pool with little turbulence, limiting the amount of suitable feeding and spawning habitat for fish and EPT invertebrates. High volumes of woody debris, plant matter and soil has accumulated around the pumping station's weed screen and settled amongst a thick layer of green algae spanning 15m in length (Fig.7.85D).

Midstream

GPS Coordinates (DD): -39.579622, 176.932111
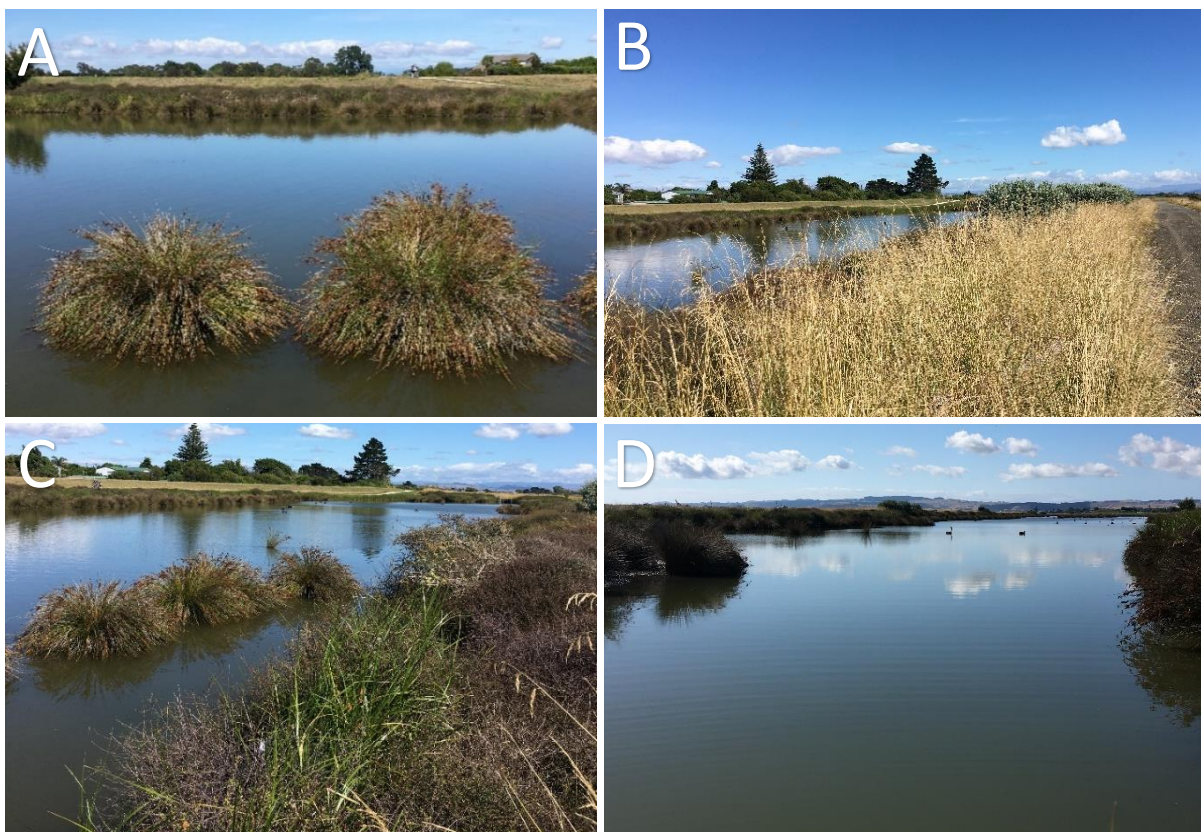
Figure 7.86. The midstream habitat along Muddy Creek. **A**: Native tussocks span the left bank and encroach into the stream. **B**: Dense exotic grasses and sparsely scattered trees line the upper reaches of the right bank. **C**: Brambles and tussocks line the lower reaches of the right bank. **D**: Little riparian shading is cast across the stream throughout the day.

As with the upstream sub-site, Muddy Creek's midstream sub-site has little hydraulic heterogeneity, although there is some diversity amongst the riparian vegetation along the right bank. Native tussocks span the 3m width of the left bank, encroaching into the waterway and providing vast amounts of suitable feeding and spawning habitat for migrating fish (Fig.7.86A). The right bank spans >7m in width and has a height of ~2m from the water's surface. Mature trees are sparsely scattered amongst dense exotic grasses along the upper reaches of the right bank (Fig.7.86B). Bramble bushes and native tussocks span the lower reaches of the bank, overhanging into the stream and providing additional suitable fish habitat (Fig.7.86C). During a high tide, a wetland habitat measuring 2m in length, forms between the right bank and a row of native tussocks, though as the flood gate opens and the tide recedes, the water level falls and a layer of thick mud is exposed in its place. The fluctuating water levels have caused scouring along the water's edge, leading to the active erosion of ~25% of each bank.

Despite the abundance of dense vegetation, shading is cast across less than 10% of the stream, exposing large amounts of the waterway to direct sunlight (Fig.7.86D). Approximately 40% of the stream is choked with dense submerged macrophytes, providing an abundance of cover and resting habitat for migrating fish. Fine sediment like sand and gravel, and small substrates like cobbles line >75% of the stream bed, though diverse hydraulic components like riffles and cascades remain sparse, limiting the amount of suitable habitat for EPT colonisation. There is a slow flow of water across the sub-site when the FFFG is closed, though the turbulence increases further downstream around the drainage structure.

Downstream

GPS Coordinates (DD): -39.575241, 176.928691
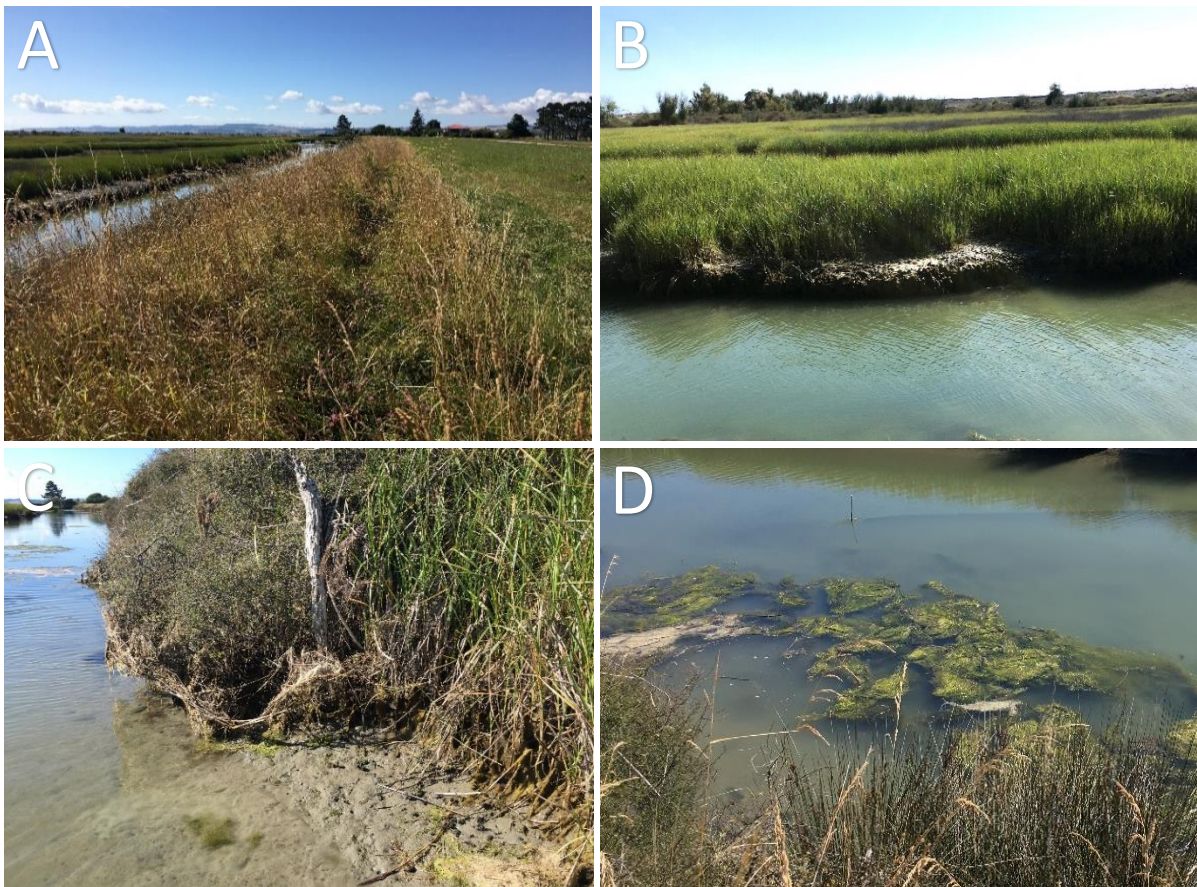
Figure 7.87. The downstream habitat along Muddy Creek. **A**: Dense long grasses, shrubs and bramble bushes span the left bank. **B**: Young grass lies above the undercut right bank. **C**: Woody debris and plant matter have become trapped within the thick mud which covers the stream bed. **D**: Floating macrophytes have become entangled within the riparian vegetation.

The downstream sub-site along Muddy Creek has little hydraulic heterogeneity, though there is some diversity amongst its riparian vegetation and substrate types. The left bank spans ~7m in width and has a height of ~3m from the water's surface. Dense long grasses line the upper reaches of the bank and a row of brambles and tussocks line the lower reaches, encroaching into the waterway and providing some suitable resting habitat for migrating fish (Fig.7.87A). Dense young grasses span the 3m width of the right bank and lie ~0.5m above the water's surface during a low tide. The daily fluctuations of the tide have scoured ~50% of each bank, causing significant active erosion in some areas. Undercut banks provide additional suitable habitat for resting fish and invertebrate species and cast some shading along the stream, though if the current rate of erosion continues, the banks will degrade further and eventually collapse into the water channel (Fig.7.87B).

At least 75% of the stream bed is covered by fine sediment like clay and silt, though large substrate types like boulders and cobbles are sparse, limiting the amount of favourable habitat for EPT colonisation. Some fallen woody debris and plant matter has accumulated along the left bank's bramble bushes, becoming trapped within the thick mud and exposed during a low tide (Fig.7.87C). Submerged macrophyte abundance is high within 40% of the stream and floating macrophytes have drifted downstream, becoming entangled within the riparian vegetation (Fig.7.87D). There is some turbulence in the furthest downstream reaches of the sub-site, though the water flow slows as the stream continues upstream.

Results

Inanga and All Species Total Abundance

H_0*: Inanga and all species total abundances will not be higher upstream, nor midstream than they are downstream when upstream migrations are restricted.*

H_1*: Inanga and all species total abundances will be higher upstream, and midstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 262.04) yields a Pearson-chisq dispersion value of 15.21 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 90.12) yields a Pearson-chisq value of 0.82 (p = 0.66) = overdispersion does not exist. An additional Vuong AIC z statistic of 10.57 21 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 3459.98) yields a Pearson-chisq dispersion value of 39.53 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 498.64) yields a Pearson-chisq statistic of 1.05 (p = 0.32) = overdispersion does not exist. An additional

Vuong AIC z statistic of -0.23 (p = 0.41) shows that the **zero-inflated negative binomial GLM** (AIC = 497.78) is favoured over a negative binomial GLM as zero-inflation occurs.

Results:

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 13) along the Muddy Creek (p = 1) (Fig.7.88) as an IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 2.17 for the DS sub-site (95% CI: 0.86 – 5.46). Wide US CIs arose due to the sub-site's small sample size. In contrast, the inanga abundance in the midstream (MS) sub-site (N = 286) is significantly higher than the inanga abundance in the DS sub-site along the Muddy Creek (p = 3.96E-07) (Fig.7.88) as an IRR value of 22.00 for the MS sub-site (95% CI: 6.66 – 72.66) is considerably higher than the IRR value of the DS sub-site. These findings show that the abundance of inanga is highest in the MS sub-site across the creek and that the FFFG dividing the MS and the DS sub-sites does not restrict upstream migrations. However, as inanga do not inhabit the US sub-site, the pumping station, which divides the US and the MS sub-sites does restrict upstream inanga migrations and therefore acts as a full barrier.

The total abundance of all species in the US sub-site (N = 28) is not significantly higher than the total abundance of all species in the DS sub-site (N = 151) along the Muddy Creek (p = 0.75) (Fig.7.88) as an IRR value of 0.71 for the US sub-site (95% CI: 0.09 – 5.67) is not higher than an IRR value of 2.52 for the DS sub-site (95% CI: 1.11 – 5.69). The total abundance of all species in the MS sub-site (N = 696) is significantly higher than the total abundance of all species in the DS sub-site along the Muddy Creek (p = 8.93E-03) (Fig.7.88) as an IRR value of 4.61 for the US sub-site (95% CI: 1.47 – 14.46) is higher than the IRR value of the DS sub-site. As with the inanga abundance analysis, these findings show that the greatest total abundance of all species was within the MS sub-site, showing that the FFFG allows full upstream passage and therefore does not act as a barrier, whereas the smallest abundance was observed within the US sub-site, suggesting that the pumping station does restrict the upstream passage of fish and therefore acts as a barrier.

The estimated odds ratio (OR) of observing an excess zero in the US or the MS data was 4656.41 times and 0.003 times, respectively, the odds of there being an excess zero in the DS data (OR = 0.0006, p = 0.92).
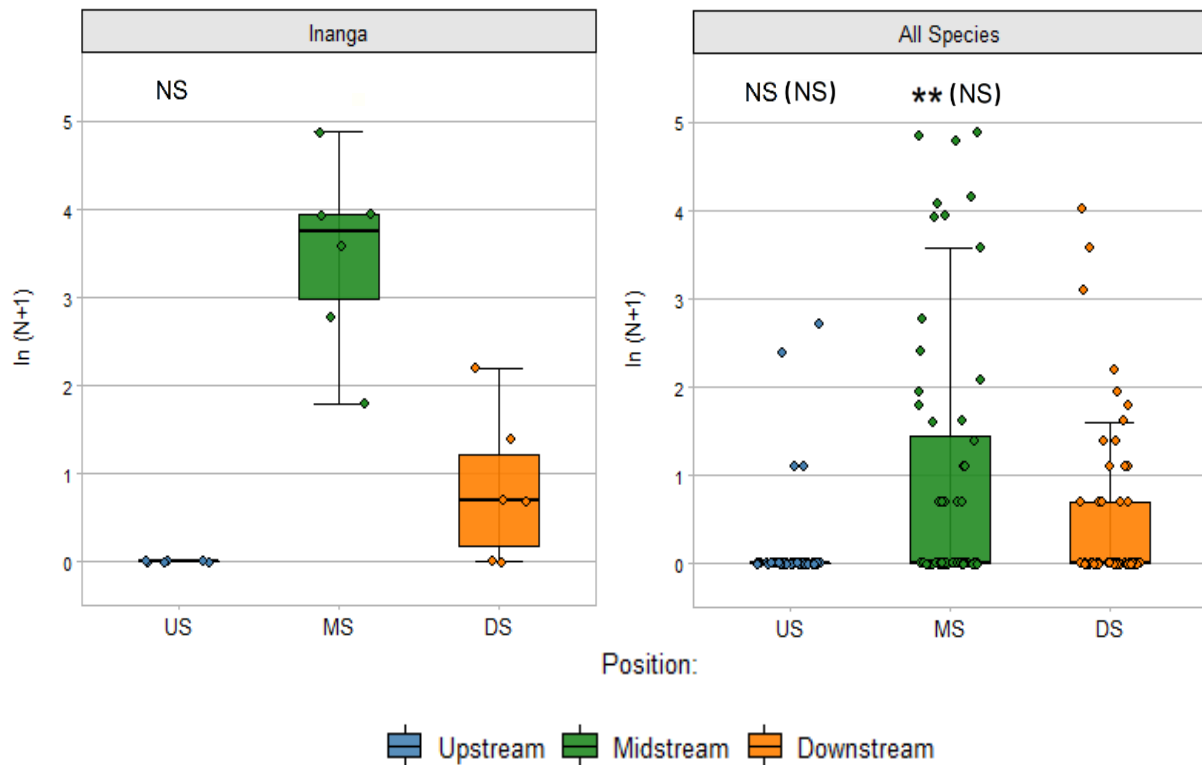
Figure 7.88. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US), midstream (MS) and downstream (DS) positions along the Muddy Creek site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance levels without brackets indicate count model p-values, significance levels within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream, and midstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream, and midstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0) along Muddy Creek is significantly different to the H' value of the DS sub-site (H' = 0.95) as an US ENS value of 1.00 (H'$_{max}$ = 0, p = 1.5E-19) and a DS ENS value of 2.58 (H'$_{max}$ = 1.79) shows that there is a greater biodiversity of species within the downstream sub-site and that the pumping station restricts upstream fish passage. However, the H' value of the MS sub-site (H' = 0.86) is not significantly different to the H' value of the DS sub-site as a MS ENS value of 2.37 (H'$_{max}$ = 1.79, p = 0.37) shows that there is a similar biodiversity of fish within both of the MS and the DS sub-sites and that the FFFG does not restrict the upstream passage of fish.

As the US sub-site yielded a H' value of 0, it was not possible to compute an US Pielou's evenness index (J'), although the US species richness (S) (S = 1) is considerably lower than the DS richness (S = 6) and a Sørensen's similarity index (ß) of 0 indicates that no species present across the site were found within both of the US and the DS sub-sites. In contrast, the species evenness is similar between the MS sub-site (J' = 0.48) and the DS sub-site (J' = 0.53), and the species richness in the MS

(S = 6) is the same as the richness within the DS sub-site (S = 6). A ß value of 0.5 indicates that 50% of the species present across the creek are found within both of the MS and the DS sub-sites. The migratory inanga and the yellow-eye mullet, and the invasive mosquitofish are the three species present within both of the MS and the DS sub-sites.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream, the midstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature measures are similar within all three sub-sites. The dissolved oxygen saturation is highest upstream and lowest midstream, whilst the specific conductivity increases further downstream. As the FFFG is closed during periods of a high tide, nutrient mixing is limited across the wide and deep MS sub-site, leading to the low observed DO saturation, although these levels remain within the threshold of inanga and have allowed large populations of inanga to thrive in abundance. The observed increasing SPC levels downstream were expected as the creek is geographically within a closer proximity to the open ocean, which generally exhibits higher specific conductivities.

7.2.12. County Drain (codr)

Site Description

<u>Date of Study:</u>

Thursday 28th November 2019

<u>Weather Conditions:</u>

Sunny, clear skies throughout the day. Little wind with warm air temperatures.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.1m at 13:47; highest tides of 1.8m at 07:33 and 19:55 (LINZ, 2020).
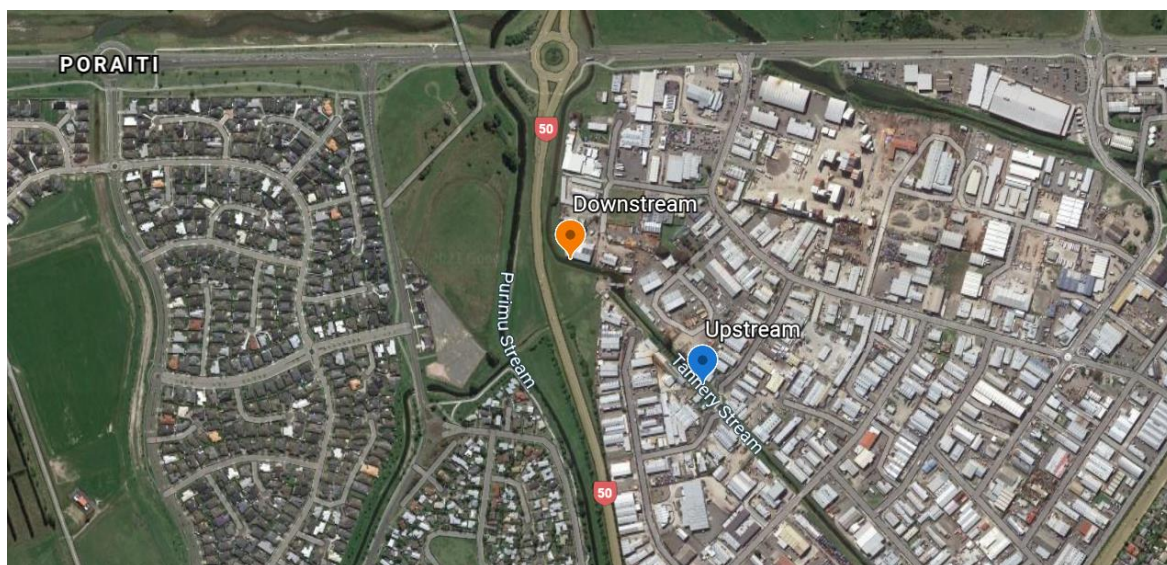
Figure 7.89. Google Earth image showing the location of the studied upstream and downstream County Drain sites along the Tannery Stream, in relation to the nearby Purimu Stream, Highway 50 and the village of Poraiti (Google Earth, 2021).

The County Drain sites lie along the Tannery Stream which branches from the Ahuriri Estuary in the north, running parallel to the Purimu Stream and State Highway 50 (Fig.7.89). The Tannery meanders into an industrial site in the village of Onekawa and continues in a south easterly direction before joining the Purimu Stream in Jervoistown. The downstream County Drain sub-site lies 3.7km from the open ocean and 5.9km of available habitat lies above the upstream sub-site.

The County Drain sites ran through public grounds; therefore, access did not require permits, landowners permissions or the use of the HBRC master key.

<u>Drainage structure present:</u>

A pumping station divides the upstream and the downstream County Drain sub-sites. Built along one of the most flood-prone areas of Hawke's Bay, the drainage structure spans ~22.5m in length and ~9.5m in width. The pumping station has a maximum flow capacity of 3.4m^3/s, though during settled climatic conditions and with the use of only one of the three jockey pumps, 0.051m^3 of water is pumped every second. Access to the pumping station was limited, hence no photographs of the structure could be taken.

GPS Coordinates (DD): -39.502015, 176.877321

Figure 7.90. The upstream habitat along County Drain. **A**: Tall trees and dense shrubs line the left bank, casting shade across large areas of the stream. **B**: Sedges, tussocks and long grass line the right bank, though the upper area of the riparian strip is heavily mown.

The upstream sub-site along County Drain has little hydraulic heterogeneity, though there is some diversity amongst the vegetation along the riparian buffers. Mature trees are sparsely scattered along the left bank between shrubs, sedges, tussocks and long grasses (Fig.7.90A). The dense vegetation spans the ~3m wide bank and encroaches into the stream, forming a small wetland habitat along the water's edge. Trees and taller shrubs cast shading across ~25% of the stream, minimising macrophyte levels along large areas of the sub-site, though the slumping of the trees has led to the active erosion of ~50% of the left bank. The right bank spans ~3m in width and has a height of ~2m from the water's surface. Sedges and tussocks are scattered amongst dense, exotic grass which overhangs and encroaches into the stream, providing cover or resting habitat for aquatic species (Fig.7.90B). The upper area along the right bank lacks diversity and is regularly and heavily mowed.

Fine sediment, like sand and clay lines ~50% of the stream bed, though larger substrate types, like boulders and cobbles are sparse. Diversity amongst hydraulic components is low as pools, riffles and cascades with fast runs and turbulence are not present, limiting the amount of habitat suitable for EPT colonisation. However, woody debris and plant matter fallen into the stream from each bank may provide alternative habitat for invertebrate species, as well as acting as suitable resting sites for resident or migratory fish.

GPS Coordinates (DD): -39.500019, 176.875057

Figure 7.91. The downstream habitat along County Drain. **A & B**: Heavily mown grass spans the width of the left bank, whilst a thin strip of long grass lines the lower reaches of the right bank, casting a small amount of shade across the stream.

The downstream County Drain sub-site has very little diversity amongst hydraulic components or riparian vegetation. The left bank spans ~2m in width and has a height of ~2m from the water's surface. Heavily mown grass spans the width of the riparian strip, casting no shade across the stream and providing no suitable habitat for fish or invertebrate species (Fig.7.91A & 7.91B). Heavily mown grass also spans much of the ~2m wide right bank, though a thin strip of long grass lines the bank's edge (Fig.7.91A). A small amount of shading is cast onto the stream from the overhanging grass strip, though macrophytes remain sparse across both shaded and unshaded areas of the sub-site. Due to the stream's slow water flow and lack of turbulence, little active or recent bank erosion was observed, though slightly undercut banks beneath the overhanging grass strip may be due to a fluctuating water level during high and low tides (Fig.7.91B).

At least 75% of the stream bed is covered by fine sediment as larger substrates are sparse. The lack of boulders or cobbles limits the amount of suitable habitat required for EPT colonisation as well as reducing the number of hydraulic components like riffles, pools or cascades. The highly modified and simplistic nature of the downstream sub-site provides little habitat for resident or migratory fish to rest, seek refuge, feed or spawn.

Results

<u>Inanga and All Species Total Abundance</u>

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

<u>Model Justification:</u>

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 215.01) yields a Pearson-chisq dispersion value of 25.35 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 50.02) yields a Pearson-chisq value of 0.66 (p = 0.77) = overdispersion does not exist. An additional Vuong AIC z statistic of 1.38E+04 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 1002.96) yields a Pearson-chisq dispersion value of 29.14 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 237.11) yields a Pearson-chisq statistic of 0.88 (p = 0.83) = overdispersion does not exist. An additional Vuong AIC z statistic of 1.78 (p = 0.04) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

<u>Results:</u>

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 102) along the County Drain (p = 1) (Fig.7.92). An IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 17.00 for the DS sub-site (95% CI: 5.09 – 56.78), showing that inanga abundances are higher downstream, and that the pumping station restricts the upstream migrations of inanga. Wide CIs arose due to the small inanga abundance sample size along the US sub-site.

The total abundance of all species in the US sub-site (N = 64) is not significantly higher than the total abundance of all species in the DS sub-site (N = 123) along the County Drain (p = 0.41) (Fig.7.92). An IRR value of 0.52 for the US sub-site (95% CI: 0.11 – 2.45) is not higher than an IRR value of 2.05 for the DS sub-site (95% CI: 0.69 – 6.09), showing that a higher total abundance of all species inhabit the downstream reaches of the stream and that the pumping station restricts upstream fish passage.
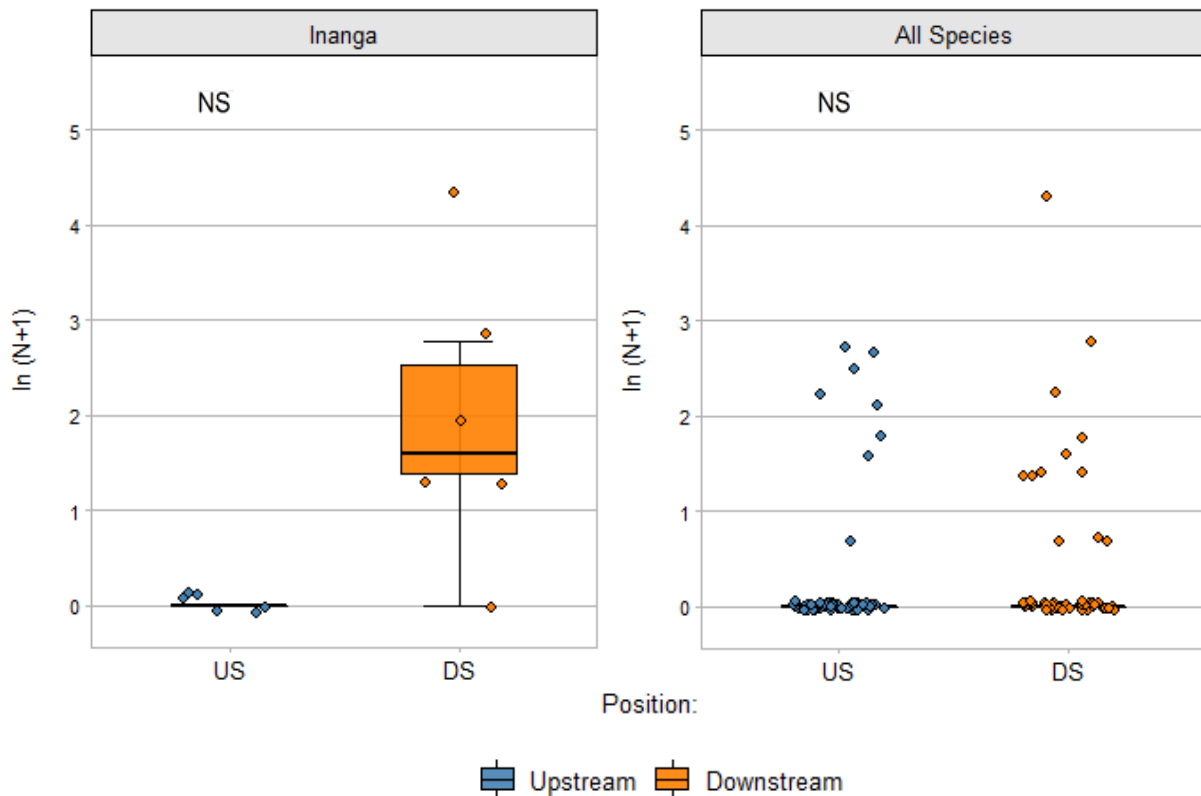
Figure 7.92. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the County Drain site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and 'All Species').

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.76) along the County Drain is not significantly different to the H' value of the DS sub-site (H' = 0.64) as an US ENS value of 2.13 (H'$_{max}$ = 1.10, p = 0.30) and a DS ENS value of 1.90 (N = 123, H'$_{max}$ = 1.39) shows that a similar biodiversity of species is present within both sub-sites.

Pielou's evenness index (J') is higher in the US sub-site (J' = 0.69) compared with the evenness in the DS sub-site (J' = 0.46), although the species richness (S) in the US (S = 3) is lower than the richness DS (S = 4). A Sørensen's similarity index (ß) of 0.29 indicates that only 29% of species present across the site are found within both sub-sites, in fact, the invasive mosquitofish is the only species present within both of the US and the DS sub-sites, whereas migratory species like inanga and yellow-eye mullet are present within the DS sub-site only. Although a similar biodiversity is observed within both sub-sites, mosquitofish and shortfin and longfin eels, which are able to physically navigate beyond a structure across terrestrial land, are the only three species inhabiting the US sub-site, and therefore the pumping station remains a full barrier to upstream migrating fish.

195

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature measures are similar within both sub-sites, although the dissolved oxygen saturation and the specific conductivity are higher downstream. As the habitat types are similar across both sub-sites, the pumping station structure alone may be the sole cause of any differences observed in water qualities between sub-sites. As the DS sub-site is unrestricted by a structure, tidal mixing and nutrient transfer across the sub-site is high, leading to a higher DO saturation and SPC level and providing suitable conditions for inanga and yellow-eye mullet to inhabit. The pumping station however also acts as an environmental barrier along the County Drain as nutrient mixing is considerably limited above the structure, leading to reduced DO and SPC levels and adverse conditions for migrating species to inhabit, but suitable conditions for invasive species to thrive in abundance.

7.2.13. Old Tutaekuri Riverbed (otri)

<u>Site Description</u>

<u>Date of Study:</u>

Thursday 5th December 2019.

<u>Weather Conditions:</u>

Clear skies and high levels of sunshine in the morning with patchy cloud cover into the afternoon; a light warm breeze throughout the day.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.4m at 06:56 rising to 1.6m by 13:13 (LINZ, 2020)

Figure 7.93. Google Earth image showing the location of the studied upstream and downstream sub-sites along the Old Tutaekuri Riverbed in relation to the nearby town of Marewa and State Highway 51 (Google Earth, 2021).

Branching from the Ahuriri Estuary and the Tannery Stream, the Old Tutaekuri Riverbed bypasses the Plantation Drain sites and continues east through the urban town of Marewa in Southern Napier (Fig.7.93). The downstream sub-site runs beneath a State Highway 51 (SH51) road bridge and lies ~5.6km from the open ocean. The upstream sub-site lies parallel to SH51 and the stream continues for ~1.4km in a south easterly direction towards the Hawke's Bay coast.

Both sub-sites run through public areas, therefore access did not require permits, permissions, or the use of the HBRC master key. Regular checking of the nets throughout the day ensured that the sites' security remained sufficient.

Drainage structure present:
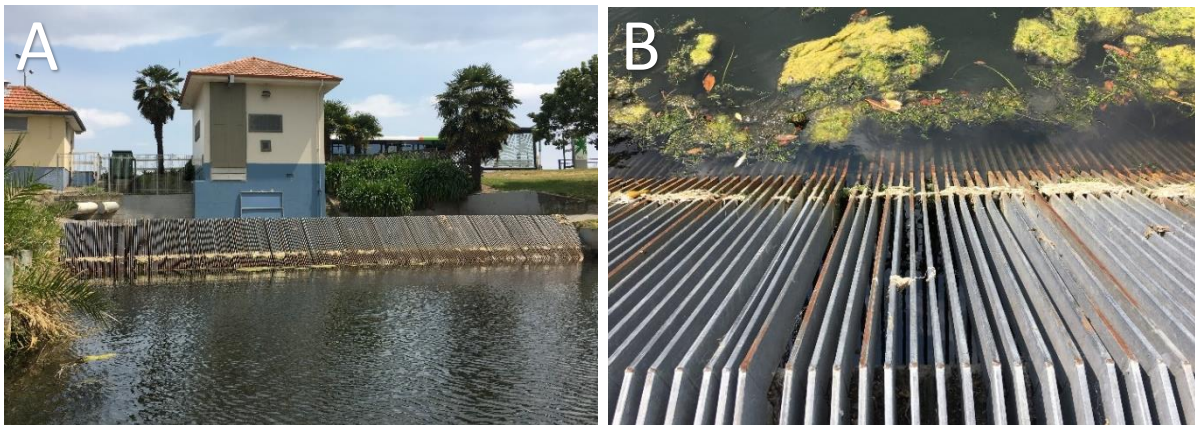
Figure 7.94. A brick-clad pumping station divides the upstream and downstream sub-sites along the Old Tutaekuri Riverbed. **A**: The structure spans the width of the waterway. **B**: A metal weed screen prevents debris from passing into the pumping station.

A brick-clad pumping station with a metal weed screen (Fig.7.94) divides the upstream and the downstream sub-sites, spanning 14m in length and 15m in width. The SH51 runs across the Old Tutaekuri Riverbed along a road bridge and behind the pumping station, spanning 47m in width.

Upstream

GPS Coordinates (DD): -39.500736, 176.905308

Figure 7.95. The upstream habitat along the Old Tutaekuri Riverbed. **A**: Heavily mown grass spans the left and the right banks. **B**: Large overhanging trees cast shading across ~25% of the stream. **C:** Fluctuating water levels have eroded the unprotected stream banks. **D**: Exposed root mats release woody debris and plant matter into the stream.

The upstream sub-site along the Old Tutaekuri Riverbed has been highly modified to become a simplistic environment with little hydraulic heterogeneity or diversity amongst its riparian vegetation. Each of the left and the right banks span ~1m in width and have been heavily mown down to the water's edge (Fig.7.95A). Mature native trees are sparsely scattered alongside each bank, casting shade across ~25% of the stream (Fig.7.95B). Submerged macrophytes are abundant in non-shaded areas, though floating macrophytes have accumulated in both shaded and non-shaded regions along both banks. Approximately 50% of each bank shows signs of active erosion as a number of large trees slump into the water channel. Fluctuating water levels during periods of high rainfall has also contributed to the erosion of the banks where there is little riparian vegetation (Fig.7.95C). Exposed root mats above the water's surface release woody debris and plant matter into the stream, providing suitable cover and resting habitat for small bodied fish (Fig.7.95D).

Larger substrate types like boulders and cobbles are sparse, limiting the amount of suitable habitat for EPT colonisation. Almost 60% of the stream bed is covered by fine deposited sediment like sand and clay, and diverse hydraulic components such as riffles, cascades and waterfalls have been replaced by simplistic pools of slow flowing water.

Downstream

GPS Coordinates (DD): -39.498282, 176.896471

Figure 7.96. The downstream habitat along the Old Tutaekuri Riverbed. **A**: Heavily mown grass spans the short riparian buffers along the left and the right banks. **B**: Mature native trees cast shading across ~40% of the stream. **C**: Overhanging trees release woody debris and plant matter into the waterway. **D**: Dense submerged macrophytes choke ~95% of the

As with the upstream sub-site, the downstream sub-site along the Old Tutaekuri Riverbed has very little hydraulic heterogeneity and minimal diversity amongst aquatic and terrestrial vegetation. Both of the left and the right riparian buffers span <0.5m in width and are heavily mown down to the water's edge (Fig.7.96A). Mature native trees are scattered along each bank, casting shade across ~40% of the stream (Fig.7.96B). A number of the overhanging trees encroach into the waterway, exposing their root mats and releasing large quantities of woody debris and plant matter into the stream, providing suitable shelter for small bodied fish and invertebrates (Fig.7.96C). Approximately 25% of each bank shows signs of active erosion due to the slumping trees and scouring at the water line as the stream's water levels rise during a high tide.

Despite the shading cast across the stream, ~95% of the waterway is choked with dense submerged macrophytes (Fig.7.96D). Additional floating macrophytes are scattered across the stream, accumulating around the banks and the exposed root mats. Although acting as suitable fish cover, this excess level of macrophyte growth substantially slows the flow of the stream, reducing the diversity of hydraulic components like riffles and cascades, and limiting any turbulence caused by the fallen woody debris. It is estimated that >75% of the stream bed is covered in fine sediment as plant matter fallen from the trees, or boulders and cobbles released from the eroding banks settles above the dense macrophyte layer, rather than settling along the stream bed.

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A suitable, well-fitting Poisson GLM (AIC = 4) yields a Pearson-chisq dispersion value of 1.23E-11 (p = 1) = overdispersion does not exist, showing that the **Poisson GLM** is the favoured model to use.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 717.36) yields a Pearson-chisq dispersion value of 11.95 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 220.64) yields a Pearson-chisq statistic of 0.52 (p = 1) = overdispersion does not exist. An additional Vuong AIC z statistic of -0.12 (p = 0.45) shows that the **zero-inflated negative binomial GLM** is favoured over a negative binomial GLM as zero-inflation occurs.

Results:

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 0) along the Old Tutaekuri Riverbed (p = 1) (Fig.7.97). The IRR value of the US sub-site is 1 times (95% CI: 0 – infinite) the IRR value of 0 (95% CI: 0 – infinite) for the DS sub-site, showing that inanga do not inhabit either sub-site along the Old Tutaekuri Riverbed and that the passability of the pumping station for upstream migrating inanga

cannot be assessed. Wide CIs arose due to the small inanga abundance sample sizes along the US and the DS sub-sites.

The total abundance of all species in the US sub-site (N = 110) is significantly higher than the total abundance of all species in the DS sub-site (N = 37) along the Old Tutaekuri Riverbed (p < 0.01) (Fig.7.97). An IRR value of 4.27 for the US sub-site (95% CI: 1.44 – 12.66) is higher than an IRR value of 2.54 for the DS sub-site (95% CI: 1.03 – 6.29), showing that a higher total abundance of all species inhabit the upstream reaches of the stream. However, additional biodiversity measures are required to assess whether these higher US abundances consist only of resident populations of invasive species, or whether native, migrating fish inhabit the US sub-site and are able to migrate beyond the structure.

The estimated odds of observing an excess zero in the US sub-site was 1.58 times the odds ratio (OR) of the DS sub-site (OR = 3.12, p = 0.03).
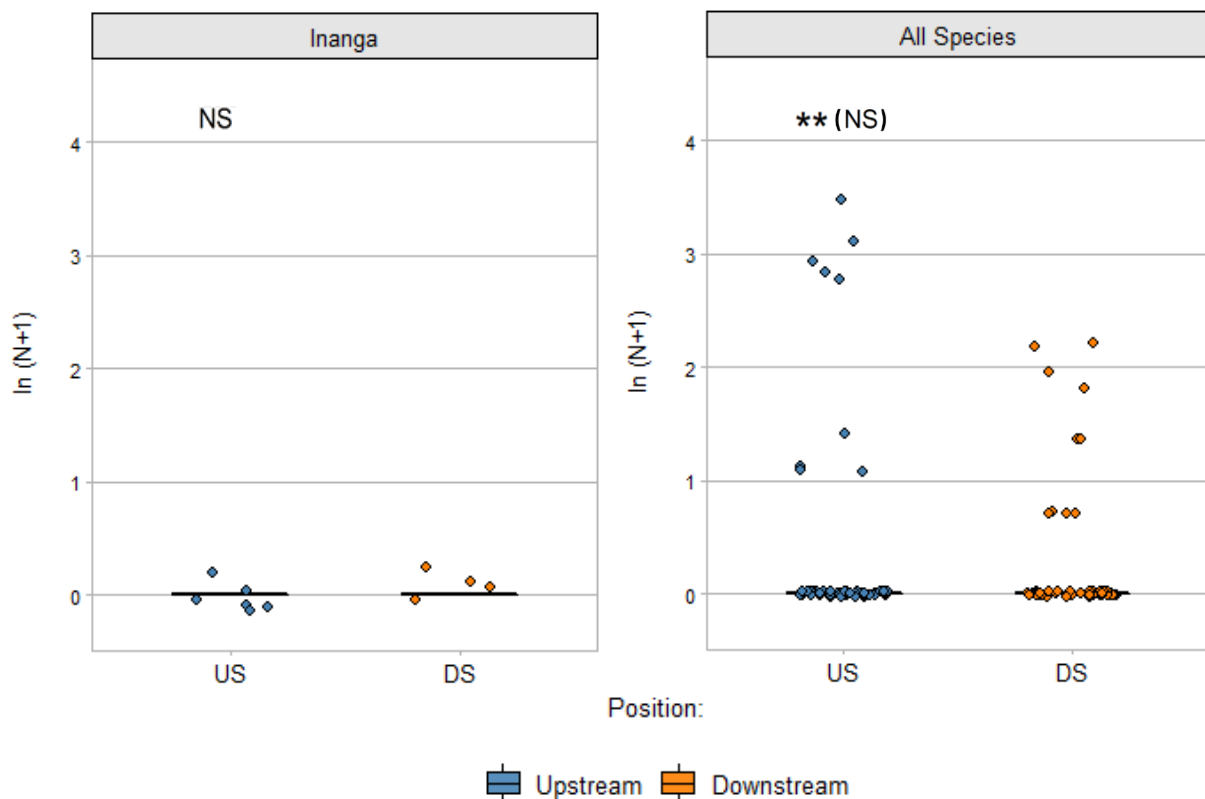
Figure 7.97. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Old Tutaekuri Riverbed. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (Poisson GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance level without brackets indicate count model p-values, significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.21) along the Old Tutaekuri Riverbed is significantly different to the H' value of the DS sub-site (H' = 0.55) as an US ENS value of 1.24 (H'_{max} = 0.69, p = 0.04) and a DS ENS value of 1.73 (H'_{max} = 1.10) shows that there is a greater biodiversity of fish inhabiting the downstream sub-site and that the pumping station does in fact restrict upstream fish passage.

Pielou's evenness index (J') and species richness (S) are lower in the US sub-site (J' = 0.31, S = 2) than the evenness and the richness in the DS sub-site (J' = 0.50, S = 3), whilst a Sørensen's similarity index of 0.80 indicates that 80% of species present across the site were found within both sub-sites. However, the invasive mosquitofish and the migratory shortfin eel are the only two species present within both of the US and the DS sub-sites, although eel sp. are able to navigate beyond barriers by physically climbing from a waterway and utilising terrestrial land. Although no migrating fish were observed DS, if populations later inhabited the DS sub-site, it is assumed that the pumping station would act as a full barrier to upstream migrating fish.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature measures are higher within the US sub-site, whilst the dissolved oxygen saturation and the specific conductivities are lower. As the habitat types are similar across both sub-sites, the pumping station structure alone may be the sole cause of any differences observed in water qualities between sub-sites. Although dense macrophytes minimise tidal exchange across much of the DS sub-site, nutrient transfer remains higher than upstream. The pumping station therefore also acts as an environmental barrier as the flow of water upstream is slow and tidal exchange is minimal, leading to lower upstream DO and SPC levels. As the stream is shallow and shading from riparian vegetation is limited, the water channel is exposed to large quantities of direct sunlight, leading to increased water temperatures and unsuitable conditions for migrating species like inanga to inhabit.

7.2.14. Purimu Stream (pust)

Site Description

<u>Date of Study:</u>

Wednesday 11th December 2019.

<u>Weather Conditions:</u>

High sunshine levels with moderate cloud throughout the day; little wind and warm air temperature.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.4m at 11:48 rising to 1.5m by 17:54 (LINZ, 2020).
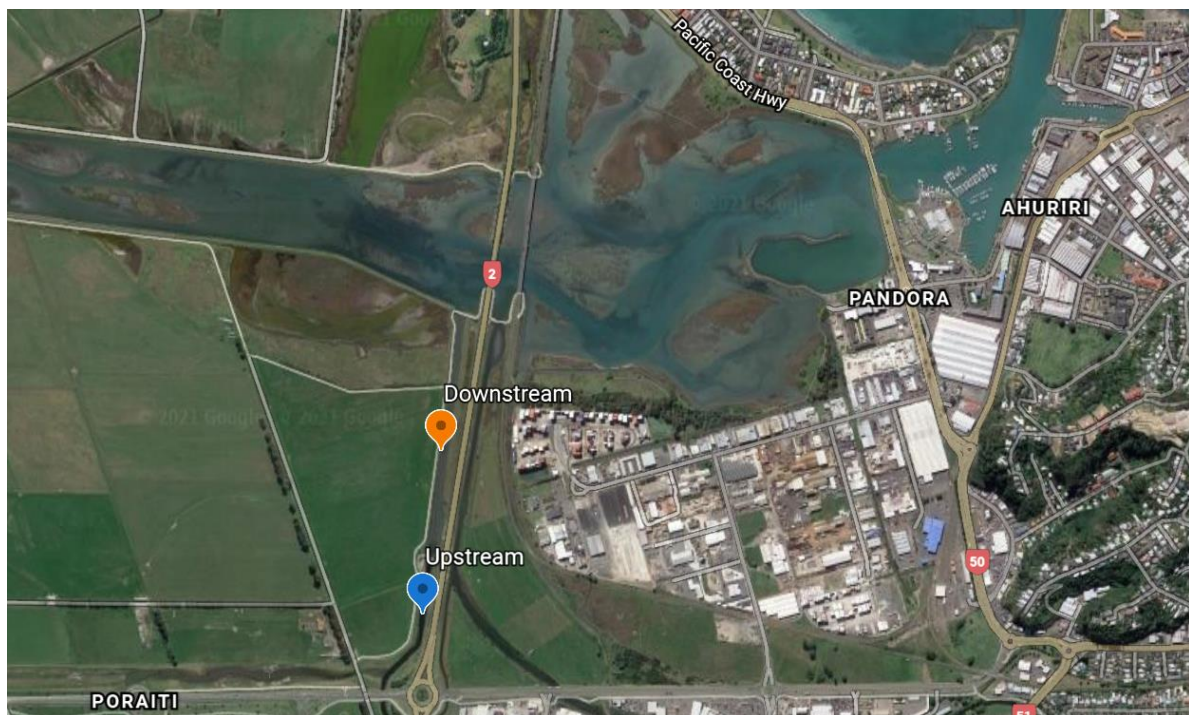
Figure 7.98. Google Earth image showing the location of the studied upstream and downstream Purimu Stream sites in relation to the nearby Ahuriri estuary, State Highway 2 and the open ocean (Google Earth, 2021).

Branching from the Ahuriri Estuary, the Purimu Stream runs in a southerly direction towards the sub-urban village of Pirimai in central Hawke's Bay. The downstream sub-site lies <3km from the open ocean and runs parallel to the Napier-Hastings Expressway (SH2) (Fig.7.98). The upstream sub-site continues south through Pirimai for >7km, before becoming the Tannery Stream in Jervoistown. At approximately 1km upstream from the site's drainage structure, a tributary named Saltwater Creek branches from the Purimu Stream which continues in a south westerly direction for an additional 2.9km, totalling at least 9.9km of available upstream habitat.

Both sub-sites ran through public areas, therefore access did not require the use of the HBRC master key and permissions were not required.

Drainage structure present:

Figure 7.99. A pumping station divides the upstream and downstream sites along the Purimu Stream. **A**: Front view showing the structure's three jockey pumps and weed/ trash screen. **B**: Rear view showing its three closed discharge culverts.

A large pumping station divides the upstream and downstream sub-sites along the Purimu Stream, measuring ~18.5m in length and ~12m in width. The structure was once considered the largest pumping station of its kind in the Southern Hemisphere due to its high flow rate. Its three jockey pumps behind the weed screen (Fig.7.99A) allow for a maximum pumping capacity of 18.4 m^3/s, though during settled climatic conditions, only one jockey pump is used, pumping 0.72m^3/s. Below the pumps, three culverts expel upstream water into the downstream habitat, though during the time of this study, the culvert outlets were closed (Fig.7.99B).

Upstream

GPS Coordinates (DD): -39.495128, 176.873899

Figure 7.100. The upstream habitat along the Purimu Stream. **A & B**: Heavily mown grass spans the width of each bank. **C**: Patches of dense aquatic grass are scattered within the stream. **D**: Submerged and floating macrophytes choke much of the waterway.

The upstream Purimu Stream sub-site has little hydraulic heterogeneity and has been transformed into a simplistic, heavily modified environment. Each of the stream banks span ~5m in width, though the left is restrained by farming fences, whilst the right is unconstrained and continues to the edge of the SH2 (Fig.7.100A & 7.100B). Heavily mown grass spans the width of each bank, casting little or no shade across the stream. Dense patches of long aquatic grass are scattered within the waterway, providing suitable shelter or feeding grounds for fish and invertebrate species (Fig.7.100C). At least 75% of the stream bed is covered by fine sediment like sand or clay, whereas larger substrates, like boulders or cobbles are sparse. The lack of diversity amongst substrates within the stream reduces the number of riffles, pools and cascades and subsequently, the amount of suitable habitat required for EPT invertebrate colonisation.

Approximately 60% of the stream is choked with hornwort *(Ceratophyllum demersum),* a submerged but often free-floating macrophyte that has formed a dense subsurface canopy (Fig.7.100D). Although offering additional habitat for some aquatic species, the introduced plant often displaces native macrophytes and provides favourable conditions for the growth of exotic or pest fish populations. The large volume of hornwort within the stream has also slowed the water flow, minimising any turbulence and therefore bank erosion.

Downstream

GPS Coordinates (DD): -39.490446, 176.874754

Figure 7.101. The downstream habitat along the Purimu Stream. **A & B**: Long grasses span the left and the right bank's riparian strip. **C**: Tree cover is sparsely scattered along each bank. **D**: Sedges and tussocks form a wetland habitat during a

As with the upstream, the downstream sub-site along the Purimu Stream has little hydraulic heterogeneity, though the habitat is much less modified and degraded. The left bank spans ~5m in width and remains constrained by farming fences, whilst the right bank spans ~10m and continues towards the edge of the SH2. Tree cover is sparsely scattered amongst long, exotic grass which spans approximately 90% of each bank (Fig.7.101A, 7.101B & 7.101C). Little or no shade is cast across the stream and macrophyte growth is minimal. Sedges and tussocks line the edges of the two banks and encroach into waterway (Fig.7.101D). During a high tide or in periods of high rainfall, the stream's water level rises so that the sedges are partially submerged within the water, creating a wetland environment and providing suitable refuge habitat for migrating fish species. The sedges also act as a buffer against the fluctuating water levels, though ~25% of the bank still shows signs of active or recent erosion.

Fine sediment covers at least 75% of the stream bed, though unlike the upstream site, small boulders and cobbles are scattered across the width of the stream. The diversity of substrate types allow for a range of hydraulic components within the waterway. During a low tide, riffles with a fast flow form over the boulders within the stream, providing suitable habitat for EPT colonisation whilst pools with a slower flow form around the banks during a high tide.

Results

NOTE: Due to the presence of an additional flood gate dividing the downstream sub-site from the main Ahuriri Estuary, downstream water levels had lowered further than had been anticipated and all six fyke nets were no longer submerged within the stream. To ensure consistency in the results, it was required that all set nets were fully submerged within a waterway for a set period of time across all studied sites. Therefore, data obtained from the downstream Purimu stream sub-site was not included in the analyses of this study. The abundances, biodiversity and water quality measures of the downstream lower Taipo stream were used to compare against the obtained data from the upstream Purimu stream sub-site.

Inanga and All Species Total Abundance

H₀: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 75.45) yields a Pearson-chisq dispersion value of 4.96 (p = 3.17E-07) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 47.62) yields a Pearson-chisq value of 0.59 (p = 0.83) = overdispersion does not exist. An additional Vuong AIC z statistic of 8.39E+06 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 1748.62) yields a Pearson-chisq dispersion value of 28.09 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 430.00)

yields a Pearson-chisq statistic of 0.86 (p = 0.87) = overdispersion does not exist. An additional Vuong AIC z statistic of -1.18 (p = 0.12) shows that the **zero-inflated negative binomial GLM** (AIC = 422.17) is favoured over a negative binomial GLM as zero-inflation occurs.

Results:

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 71) along the Purimu stream (p = 1) (Fig.7.102). An IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 11.83 for the DS sub-site (95% CI: 6.01 – 23.28), showing that inanga abundances are higher downstream, and that the pumping station restricts the upstream migrations of inanga. Wide CIs arose due to the small inanga abundance sample size along the US sub-site.

The total abundance of all species in the US sub-site (N = 255) is significantly higher than the total abundance of all species in the DS sub-site (N = 201) along the Purimu stream (p = 0.03) (Fig.7.102). An IRR value of 3.11 for the US sub-site (95% CI: 1.10 – 8.77) is higher than an IRR value of 4.02 for the DS sub-site (95% CI: 1.58 – 10.23), showing that a higher total abundance of all species inhabit the upstream reaches of the stream, although further biodiversity analyses are required to assess whether these higher abundances comprise of resident populations of invasive species, or whether native, migrating fish inhabit the US sub-site and are able to navigate beyond the pumping station.

The odds ratio (OR), or the estimated odds of observing an excess zero, in the US sub-site is 9.71 times the odds of the DS (OR = 0.20, p = 0.26) (Fig.7.102).
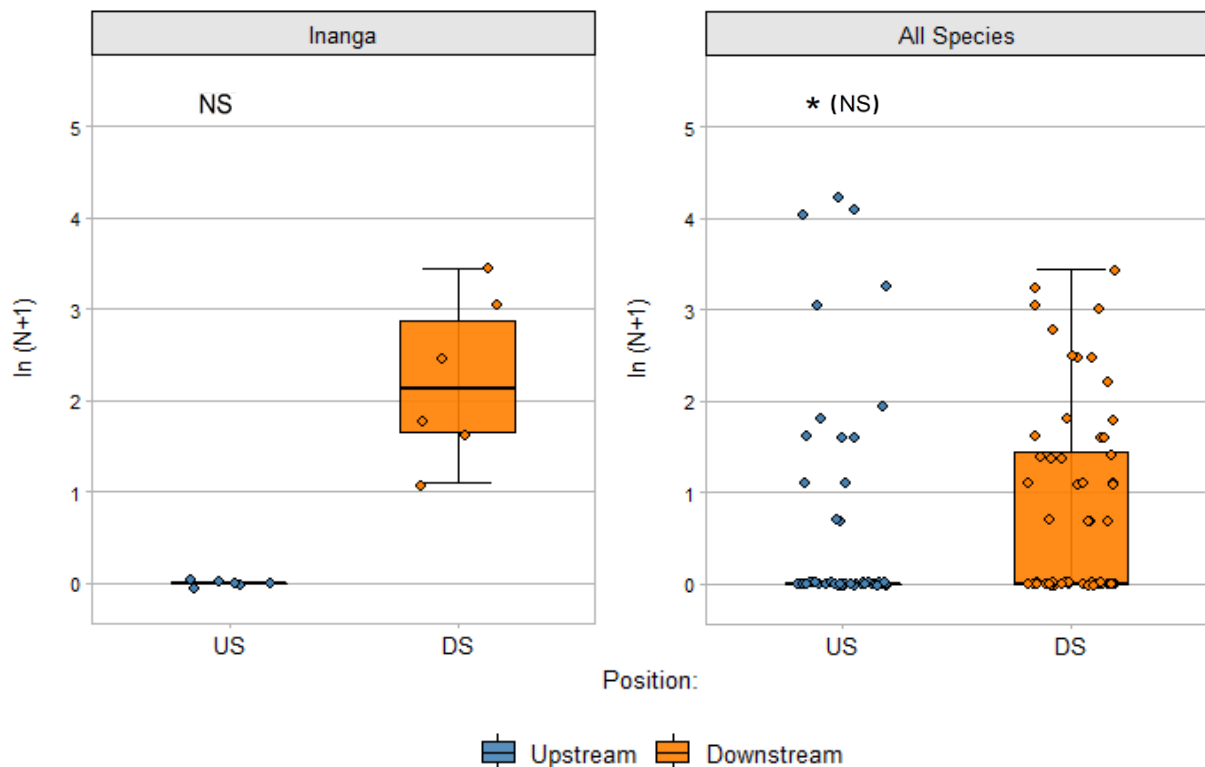
Figure 7.102. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Purimu Stream. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance level without brackets indicate count model p-values, significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H₀: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H₁: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.47) along the Purimu Stream is significantly different to the H' value of the DS sub-site (H' = 1.55) as an US ENS value of 1.59 (H'_{max} = 1.10, p = 1.70E-44) and a DS ENS value of 4.71 (H'_{max} = 1.95) shows that there is a considerably greater biodiversity of species downstream than there is upstream and that the pumping station restricts upstream fish passage.

Pielou's evenness index (J') and species richness (S) is lower in the US sub-site (J' = 0.42, S = 3) than the evenness and the richness in the DS sub-site (J' = 0.80, S = 7), whilst a Sørensen's similarity index of 0.40 indicates that only 40% of species caught across the stream are found within both sub-sites. The invasive mosquitofish and the shortfin eel are the only two shared species present in both of the US and the DS sub-sites, whereas other migratory species like inanga, grey mullet and yellow-eye mullet are present within the DS sub-site only. This shows that although the total abundance of all species is higher upstream, the pumping station still restricts upstream passage for native upstream migrating fish and therefore remains a full barrier.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, the dissolved oxygen saturation and the specific conductivity measures are all higher downstream than they are within the upstream sub-site. As observed with the County Drain site, the pumping station acts as an environmental barrier, limiting the amount of tidal mixing and nutrient transfer upstream, leading to decreased DO and SPC levels; conditions which are unfavourable for native migrating fish but are suitable for resident populations of invasive species that are more tolerant to harsher environmental conditions. Although the temperature is lower upstream, it remains high across both sub-sites due to the lack of shading cast by riparian vegetation and the shallow nature of each water channel, though temperatures remain within the thresholds for inanga to inhabit.

7.2.15. Ahuriri Station Drain (asdr)

<u>Site Description</u>

<u>Date of Study:</u>

Upstream: Friday 20th December 2019

Downstream: Friday 13th December 2019

Note: Due to the location of the site's drainage structure, and the inability to set netting within the Ahuriri Estuary, the lower Taipo Stream's downstream sub-site will also be used as the Ahuriri Station Drain's downstream sub-site. Any environmental differences between the two studied days will be taken into consideration.

<u>Weather Conditions:</u>

Upstream: Dense cloud cover throughout the day leading to heavy downpours in the afternoon; air temperature remained mild.

Downstream: Dense cloud in the morning changing to clear skies and high sunshine levels by the afternoon; increasing air temperatures through the day.

<u>Hydrographic Tide Predictions:</u>

Upstream: Tidal site. Lowest tide of 0.2m at 06:48 rising to 1.8m by 13:04 (LINZ, 2020).

Downstream: Tidal site. Lowest tide of 0.3m at 13:22 rising to 1.6m by 19:30 (LINZ, 2020)

Figure 7.103. Google Earth image showing the location of the studied upstream and downstream Ahuriri Station Drain sub-sites in relation to the villages of Ahuriri and Poraiti, the Ahuriri Estuary and the open ocean (Google Earth, 2021).

Branching north from the Ahuriri Estuary, the upstream sub-site meanders in a north-westerly direction and runs parallel to Pump Road. The sub-site lies ~0.6km west of the Hawke's Bay airport, continuing north for >8.5km past the village of Bay View in northern Hawke's Bay. The downstream sub-site branches south from the Ahuriri Estuary along the lower Taipo Stream, lying west of the village of Poraiti and 5.93km from the open ocean (Fig.7.103). Both sub-sites run through rural agricultural land.

To access the upstream sub-site, permissions were required from the state-owned farming company, 'Landcorp Farming Ltd.', who owned the land which the stream runs through. After expressing an interest in using the site and explaining the study's aims via email to the site's General Manager, it was required that a 'Risk Assessment' briefing video be watched and a detailed 'Health and Safety' talk be attended in person. Official wavery forms were signed and access was granted. It was further required that an app be installed onto a smartphone, which must be used to sign in and out of the site every time access was required.

Access to the downstream sub-site required the use of the HBRC master key. Caution was essential whilst driving a vehicle along the publicly accessible cycle path to avoid any present dangers. Regular checking of the nets throughout the day along the sub-site ensured that security remained uncompromised.

Drainage structure present:

Figure 7.104. **A:** A pumping station divides the upstream and downstream sub-sites along the Ahuriri Station Drain. **B**: A second smaller pumping station lies to the left of Ahuriri Station, though this structure will not be assessed in this study.

A large pumping station, measuring 17m in length and 10m in width, divides the upstream and the downstream Ahuriri Station Drain sub-sites (Fig.7.104A). An additional smaller pumping station lies to the left of the main structure and divides the Ahuriri Estuary from an unnamed stream which runs in a north-easterly direction (Fig.7.104B). The two structures share four pumps which suck ~17,000L of seawater per minute from the Ahuriri Estuary, though the larger of the pumping stations is the structure that will be assessed in this study.

GPS Coordinates (DD): -39.48048, 176.859032

Figure 7.105. The upstream habitat along the Ahuriri Station Drain site. **A:** Mature, overhanging trees along the left bank cast shading across the width of the stream. **B**: Dense, long grasses span the right bank. **C**: Large volumes of woody debris and plant matter has fallen into the stream. **D**: Floating macrophytes, periphyton and algae has accumulated within the fallen woody debris.

The lentic upstream sub-site along the Ahuriri Station Drain has very little hydraulic heterogeneity, turbulence is minimal and the water table rarely fluctuates. A diverse range of mature native trees span approximately 80% of the left bank which overhang across large areas of the stream, providing copious amounts of refuge habitat for aquatic species (Fig.7.105A). Beneath the canopy of mature trees, the understorey is bare and impervious, although young shrubs and long grasses are abundant along an area of the left bank which is without any tree cover. Dense, young grasses span the 5-7m wide right bank, which encroach into the stream and provides some additional fish spawning habitat (Fig.7.105B). Shading is cast across ~70% of the waterway, limiting much of the stream to direct sunlight exposure, though numerous gaps within the left bank's tree canopy has allowed sunlight to penetrate through and diffuse into the stream. This has subsequently led to the excessive and uncontrolled growth of submerged macrophytes which choke large areas of the water channel and limits the amount of suitable habitat available for EPT invertebrate colonisation.

Although the high volume of submerged macrophytes have slowed the stream's water flow significantly, active erosion has still occurred along ~50% of each bank due to fluctuating water levels during periods of high rainfall. The overhanging trees have also led to the slumping of the left bank which has resulted in the exposure of the trees' root mats and the release of large quantity of cobbles, woody debris and plant matter into the stream (Fig.7.105C). Floating macrophytes,

periphyton and algae has accumulated within the woody debris, leading to a poorer water quality across the sub-site and potentially allowing for an increase in invasive fish abundances (Fig.7.105D).

<u>Downstream (Poraiti Lane)</u>

Note: Refer to 'Downstream (Poraiti Lane)' section in the Lower Taipo Stream site description for the sub-site GPS coordinates, site images and full habitat description.

Results

Note: Only five fyke nets were set within the upstream sub-site as water levels were too deep in some areas for efficient net setting.

<u>Inanga and All Species Total Abundance</u>

H_0: Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.

H_1: Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.

<u>Model Justification:</u>

Inanga abundance analysis:

A poorly fitting Poisson GLM (AIC = 75.45) yields a Pearson-chisq dispersion value of 5.51 (p = 1.29E-7) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 47.62) yields a Pearson-chisq value of 0.65 (p = 0.75) = overdispersion does not exist. An additional Vuong AIC z statistic of 8.1E+06 (p < 2.2E-16) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 4538.13) yields a Pearson-chisq dispersion value of 91.19 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 458.26) yields a Pearson-chisq statistic of 0.79 (p = 0.95) = overdispersion does not exist. An additional Vuong AIC z statistic of -1.12 (p = 0.13) shows that the **zero-inflated negative binomial GLM** (AIC = 450.72) is favoured over a negative binomial GLM as zero-inflation occurs.

<u>Results:</u>

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 71) along the Ahuriri Station Drain (p = 1) (Fig.7.106). An IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 11.83 for the DS sub-site (95% CI: 6.01 – 23.28), showing that inanga abundances are higher downstream, and that the pumping station restricts the upstream migrations of inanga. Wide CIs arose due to the small inanga abundance sample size along the US sub-site.

The total abundance of all species in the US sub-site (N = 881) is significantly higher than the total abundance of all species in the DS sub-site (N = 201) along the Ahuriri Station Drain (p = 1.32E-05) (Fig.7.106). An IRR value of 11.97 for the US sub-site (95% CI: 3.92 – 36.55) is considerably higher than an IRR value of 3.35 for the DS sub-site (95% CI: 1.92 – 5.86), showing that a higher total abundance of all species inhabit the upstream reaches of the stream, although further biodiversity

analyses are required to assess which species inhabit the upstream sub-site and whether any upstream migrating fish are able to navigate beyond the pumping station.

The estimated odds of obtaining an excess zero in the US data is 4779 times the odds ratio (OR) of the DS data (OR = 0.0003, p = 0.9) (Fig.7.106).
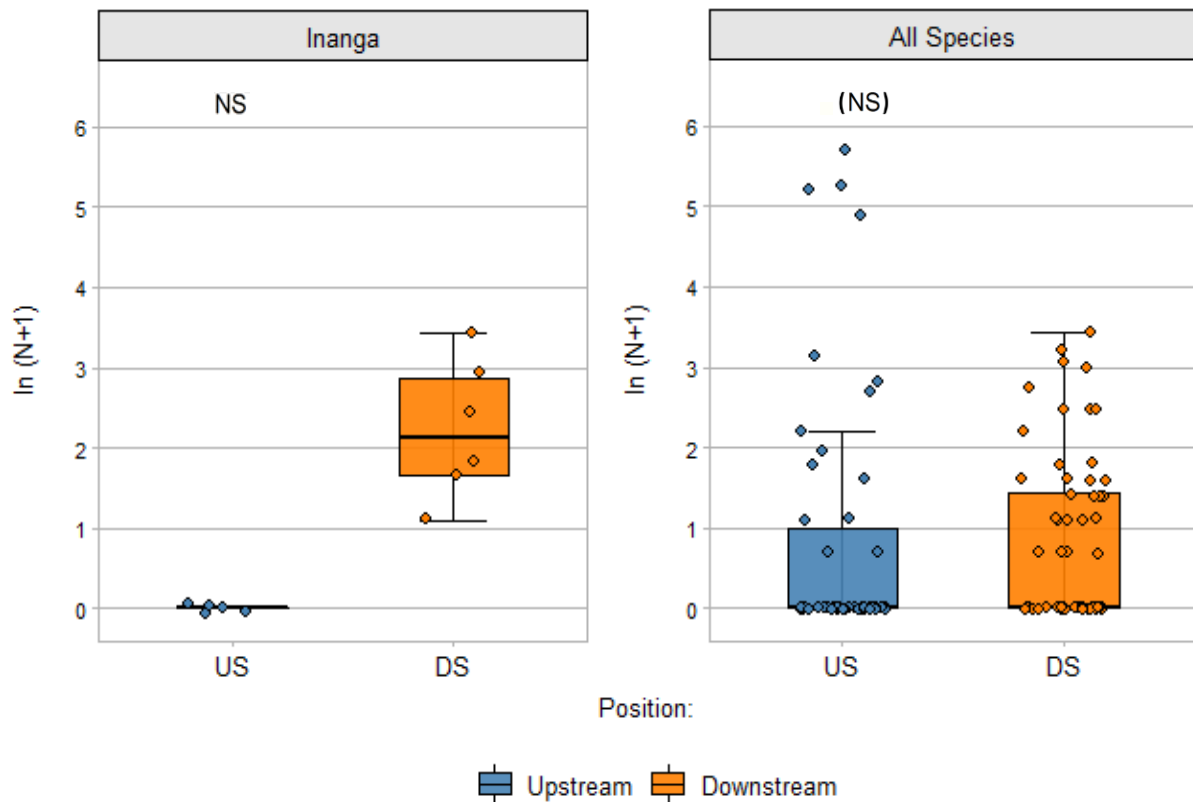
Figure 7.106. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Ahuriri Station Drain site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (negative binomial GLM for 'Inanga' and zero-inflated negative binomial GLM for 'All Species'). 'All Species' significance level without brackets indicate count model p-values; significance level within brackets indicate zero-inflated component p-values.

Species Diversity and Similarity

H_0: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H_1: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.32) along the Ahuriri Station Drain is significantly different to the H' value of the DS sub-site (H' = 1.55) as an US ENS value of 1.38 (H'$_{max}$ = 1.39, p = 6.6E-63) and a DS ENS value of 4.71 (H'$_{max}$ = 1.95) shows that there is a greater biodiversity of fish within the downstream sub-site and that the pumping station restricts upstream fish passage.

Pielou's evenness index (J') and species richness (S) is considerably lower in the US sub-site (J' = 0.23, S = 4) than the evenness and the richness within the DS sub-site (J' = 0.80, S = 7), whilst a Sørensen's similarity index of 0.55 indicates that only 55% of species caught across the waterway are found within both of the US and the DS sub-sites. Three species are present within both sub-sites: the resident common bully, the New Zealand freshwater shrimp and the invasive mosquitofish, whereas migratory species like inanga, grey mullet and yellow-eye mullet are present within the DS sub-site only, showing that the pumping station does act as a full barrier to upstream migrating fish and that the higher upstream total abundance comprises of populations of resident or invasive species only.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature, the dissolved oxygen saturation and the specific conductivity are all higher downstream than they are within the upstream sub-site. The pumping station along the Ahuriri Station Drain also acts as an environmental barrier, reducing the level of tidal exchange and nutrient transfer across the stream, leading to the lower observed DO and SPC levels. A higher macrophyte density within the US water channel has provided the optimal habitat conditions for invasive species to thrive, which has subsequently led to reduced DO saturations. A greater amount of shading cast across the US sub-site from the riparian vegetation has led to reduced exposure to direct sunlight and lower water temperatures than within the DS sub-site, which may provide favourable conditions for inanga to inhabit if upstream passage across the pumping station were possible.

7.2.16. Plantation Drain (pldr)

<u>Site Description</u>

<u>Date of Study:</u>

Wednesday 18th December 2019.

<u>Weather Conditions:</u>

Dense cloud cover in the morning changing to a heavy rain shower by the afternoon; mild air temperature remained throughout the day.

<u>Hydrographic Tide Predictions:</u>

Tidal site. Lowest tide of 0.2m at 04:59 rising to 1.8m by 11:14 (LINZ, 2020).
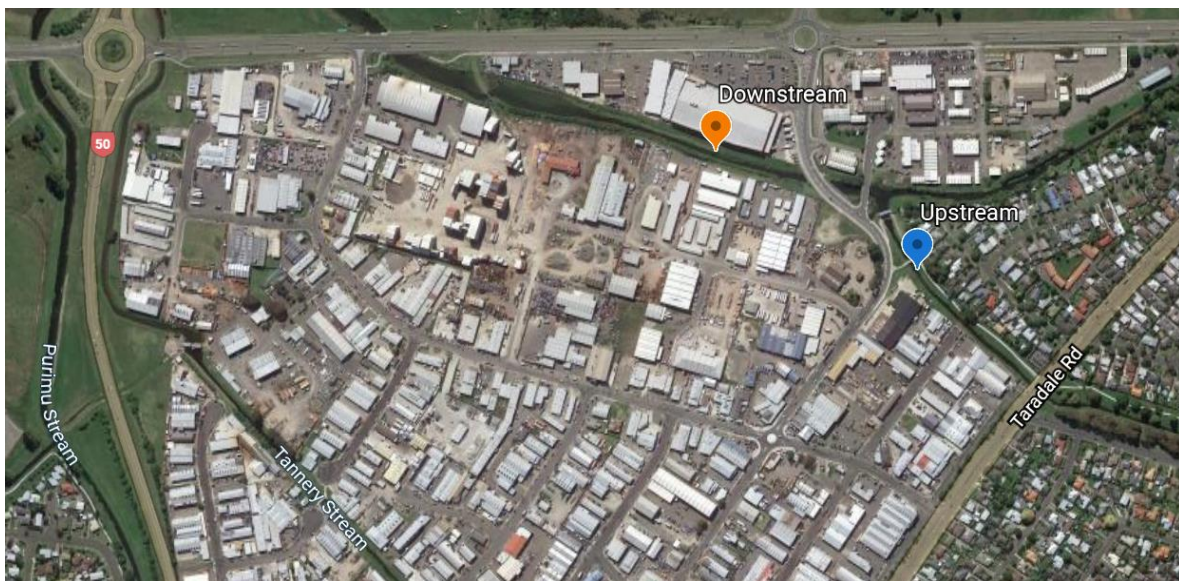
Figure 7.107. *Google Earth image showing the location of the studied upstream and downstream Plantation Drain sub-sites in relation to the Purimu and Tannery Streams, Highway 50 and Taradale Road (Google Earth, 2021).*

The Plantation Drain sites lie along an unnamed tributary of the Tannery Stream, which itself branched south from the Ahuriri Estuary. The downstream sub-site runs parallel to the Mitre 10 superstore in an easterly direction and lies ~4km from the open ocean. The tributary runs beneath Ford Road and continues upstream, branching into two smaller waterways: the first running east and becoming the Old Tutaekuri Riverbed and the second running in a south easterly direction through the Onekawa industrial estate, where the upstream sub-site lies (Fig.7.107). Approximately 3km of available habitat lies above the upstream sub-site.

Both sub-sites ran through public areas, therefore access did not require permits, landowner permissions or the use of the HBRC master key.

Drainage structure present:

Figure 7.108. **A**: A pumping station divides the upstream and downstream sub-sites along Plantation Drain. **B**: A large box culvert beneath the Ford Road bridge lies 100m downstream of the pumping station.

A pumping station divides the upstream and downstream sub-sites along Plantation Drain, spanning 11.9m in length and 10.5m in width (Fig.7.108A). Since the structure is situated within an industrial estate, a large amount of litter, plant and woody debris has accumulated around the weed screen. A large box culvert spanning 35m in length and 8m in width lies 100m downstream of the pumping station, beneath the Ford Road bridge (Fig.7.108B). Following an observational assessment, the culvert was deemed to not act as a barrier and hence was not the structure assessed in this study.

Upstream

GPS Coordinates (DD): -39.499278, 176.887345

Figure 7.109. The upstream habitat along Plantation Drain. **A**: Heavily mown grass spans the widths of the left and the right bank. **B**: Little shading is cast across the stream. **C**: Significant erosion has caused landslips along the right bank. **D**: Several pipes leak unknown liquids into the waterway.

The upstream Plantation Drain sub-site has very little hydraulic heterogeneity and the vegetation along each riparian buffer lacks diversity. Both the left and the right banks span ~4m in width and have heights of 1.5-2m from the water's surface. Heavily mown grass spans ~80% of the left bank, above a row of partially submerged sedges which line the bank's edge, providing suitable resting and spawning habitat for fish species (Fig.7.109A). Any vegetation along the width of the right bank has been heavily mown down to the water's edge, casting a small amount of shading across the stream (Fig.7.109B). Approximately 50% of the right bank shows signs of significant active erosion as large landslips have released copious amounts of soil, rock and plant matter into the waterway (Fig.7.109C).

Despite the unnatural accumulation of various substrate types beneath the landslips, favourable conditions for EPT colonisation remain minimal. The slow water flow along the highly modified and simplistic sub-site is due to the lack of riffles, cascades and waterfalls along the stream. Submerged and floating macrophytes are sparse, limiting the amount of suitable habitat for aquatic species. The water quality across the sub-site is poor and a large amount of public litter has accumulated around the pumping station's screen. Several drainage pipes leak unknown, oil-based liquids directly into the stream, polluting the water and killing surrounding vegetation (Fig.7.109D).

Downstream

GPS Coordinates (DD): -39.497763, 176.884055

Figure 7.110. The downstream habitat along Plantation Drain. **A**: Native flaxes, sedges and small trees span the upper reaches of the left bank. **B**: Mature native trees cast shade across the stream from the lower reaches of the left bank. Heavily mown grass lines the lower reaches of the right bank. **C**: Dense shrubs and exotic grasses span the upper reaches of the right bank. **D**: A slow flow across the site is due to the high level of macrophytes which choke the water channel.

The downstream Plantation Drain sub-site has little hydraulic heterogeneity, though the riparian vegetation and substrate types along the stream are diverse. Native flaxes, sedges and small trees are scattered amongst grasses of varying height along the upper reaches of the left bank. The vegetation spans the 5m riparian buffer and encroaches into the stream, offering resident or migrating fish suitable resting and spawning habitat (Fig.7.110A). Along the lower reaches of the left bank, small shrubs are replaced by mature native trees with a diverse and intact understorey. The large overhanging trees cast a significant amount of shade across the stream and provide additional habitat for aquatic species along the bank's edge (Fig.7.110B). Dense shrubs and exotic grasses span the 5m width of the right bank's upper reach, whilst encroaching into the stream, forming a wetland habitat during a high tide (Fig.7.110C). As the right bank continues downstream, the dense vegetation is replaced by a heavily mown and degraded buffer, spanning <1m in width (Fig.7.110B). Little shading is cast onto the stream from the right bank and the lack of vegetation acting as a defence has led to the recent erosion of ~35% of the bank.

Diverse hydraulic components like riffles, cascades and waterfalls are minimal along the stream, reducing the amount of suitable habitat for EPT colonisation. Despite this, boulders and cobbles are abundant onsite, particularly beneath the recently eroded areas of the right bank. Approximately half of the stream bed is covered by fine sediment, with woody debris and leaf matter constituting the remaining 50%, providing suitable fish cover habitat. Dense macrophytes choke ~75% of the waterway, though are most abundant in non-shaded areas, slowing the water flow and limiting turbulence across the sub-site (Fig.7.110D).

Results

Inanga and All Species Total Abundance

H_0: *Inanga and all species total abundances will not be higher upstream than they are downstream when upstream migrations are restricted.*

H_1: *Inanga and all species total abundances will be higher upstream than they are downstream when upstream migrations are unrestricted.*

Model Justification:

Inanga abundance analysis:

A suitable, well-fitting Poisson GLM (AIC = 14.16) yields a Pearson-chisq dispersion value of 0.3 (p = 0.98) = overdispersion does not exist, showing that the **Poisson GLM** is the favoured model to use.

All Species total abundance analysis:

A poorly fitting Poisson GLM (AIC = 467.57) yields a Pearson-chisq dispersion value of 7.93 (p < 2.2E-16) = overdispersion exists. A more suitable, better fitting negative binomial GLM (AIC = 231.57) yields a Pearson-chisq statistic of 0.91 (p = 0.76) = overdispersion does not exist. An additional Vuong AIC z statistic of 0.27 (p = 0.39) shows that the **negative binomial GLM** is favoured over a zero-inflated negative binomial GLM as zero-inflation does not occur.

Results:

The inanga abundance in the upstream (US) sub-site (N = 0) is not significantly higher than the inanga abundance in the downstream (DS) sub-site (N = 3) along the Plantation Drain (p = 1) (Fig.7.111). An IRR value of 0 for the US sub-site (95% CI: 0 – infinite) is not higher than an IRR value of 0.50 for the DS sub-site (95% CI: 0.16 – 1.55), showing that inanga abundances are higher downstream, and that the pumping station restricts the upstream migrations of inanga. Wide CIs arose due to the small inanga abundance sample size along the US sub-site.

The total abundance of all species in the US sub-site (N = 38) is not significantly higher than the total abundance of all species in the DS sub-site (N = 59) along the Plantation Drain (p = 0.47) (Fig.7.111). An IRR value of 0.64 for the US sub-site (95% CI: 0.20 – 2.10) is not higher than an IRR value of 0.98 for the DS sub-site (95% CI: 0.43 – 2.24), showing that a higher total abundance of all species inhabit the downstream reaches of the stream and that the pumping station restricts upstream fish passage.

Figure 7.111. Box and whisker and jitter plots of ln(N+1) transformed inanga and all species total abundances against upstream (US) and downstream (DS) positions along the Plantation Drain site. DS was the reference group for the position independent variable. Significance levels: *** represents p-values < 0.001, ** < 0.01, * < 0.05, NS > 0.05 (Poisson GLM for 'Inanga' and negative binomial GLM for 'All Species').

Species Diversity and Similarity

H$_0$: Biodiversity, species richness and species evenness upstream will not differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is unrestricted.

H$_1$: Biodiversity, species richness and species evenness upstream will differ to the biodiversity, species richness and species evenness downstream if upstream fish passage is restricted.

The Shannon Weiner diversity index (H') of the US sub-site (H' = 0.69) along the Plantation Drain is significantly different to the H' value of the DS sub-site (H' = 1.35) as an US ENS value of 2.00 (H'$_{max}$ = 0.69, p = 2.20E-06) and a DS ENS value of 3.84 (H'$_{max}$ = 1.95) shows that there is a greater biodiversity of fish inhabiting the downstream sub-site than there is upstream, and that the pumping station restricts upstream fish passage.

Pielou's evenness index (J') is higher in the US sub-site (J' = 1.00) than the evenness within the DS sub-site (J' = 0.69), although the US species richness (S) (S = 2) is considerably lower than the richness DS (S = 7). A Sørensen's similarity index of 0.44 indicates that only 44% of species present across the site are found within both sub-sites. The invasive mosquitofish and the shortfin eel are the only two species present within both of the US and the DS sub-sites, whereas migratory species like inanga and yellow-eye mullet are present within the DS sub-site only.

Environmental Effect on Species Abundance

The mean water qualities within each of the upstream and the downstream sub-sites, presented within the water quality comparison table in the main text, indicate that the mean temperature measures are similar within both sub-sites, although the dissolved oxygen saturation and the specific conductivity are higher downstream. As the habitat types are similar across both sub-sites, the pumping station structure alone may be the sole cause of any differences observed in water qualities between sub-sites. As with the previous waterways where a physical barrier is present, an environmental barrier forms between the two sub-sites, where lower upstream DO and SPC levels are caused by reduced tidal mixing and nutrient transfer. The altered water qualities provide the optimal conditions for more environmentally tolerant, invasive species to inhabit, though unsuitable conditions for native species to inhabit if upstream passage were possible.

Chapter 8
References

Abell, R., Thieme, M., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S., Bussing, W. and Stiassny, M. 2008. Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. *BioScience*. **58,** 403-414.

Allibone, R. 1999. Impoundment and introductions: their impacts on native fish of the upper Waipori River, New Zealand. *Journal of the Royal Society of New Zealand*. **29,** 291-299.

Anderson, G., Freeman, M., Freeman, B., Straight, C., Hagler, M. and Peterson, J. 2012. Dealing with uncertainty when assessing fish passage through culvert road crossings. *Environmental Management*. **50,** 462-477.

Anderson, J., Ward, R., Petty, J., Kite, J. and Strager, M. 2014. Culvert effects on stream and stream-side salamander habitats. *International Journal of Environmental Science and Development*. **5,** 274.

Araújo, C., Silva, D., Gomes, L., Acayaba, R., Montagner, C., Moreira-Santos, M., Ribeiro, R. and Pompêo, M. 2018. Habitat fragmentation caused by contaminants: atrazine as a chemical barrier isolating fish populations. *Chemosphere.* **193,** 24-31.

Attrill, M. and Rundle, S. 2002. Ecotone or ecocline: ecological boundaries in estuaries. *Estuarine, Coastal and Shelf Science.* **55,** 929-936.

Auster, P. 1998. A conceptual model of the impacts of fishing gear on the integrity of fish habitats. *Conservation Biology.* **12,** 1198-1203.

Baker, C. 2003. Effect of fall height and notch shape on the passage of inanga *(Galaxias maculatus)* and common bullies *(Gobiomorphus cotidianus)* over an experimental weir. *New Zealand Journal of Marine and Freshwater Research*. **37,** 283-290.

Baker, C. and Boubée, J. 2006. Upstream passage of inanga *Galaxias maculatus* and redfin bullies *Gobiomorphus huttoni* over artificial ramps. *Journal of Fish Biology*. **69,** 668-681.

Behlke, C., Kane, D., McLean, R. and Travis, M. 1991. *Fundamentals of culvert design for passage of weak-swimming fish*. [Online]. Fairbanks, AK: Alaska Department of Transportation and Public Facilities. [Accessed 22 October 2021]. Available from: https://scholarworks.umass.edu/fishpassage_reports/291/

Belliard, J., Beslagic, S., Delaigue, O. and Tales, E. 2018. Reconstructing long-term trajectories of fish assemblages using historical data: The Seine River basin (France) during the last two centuries. *Environ. Sci. Pollut. Res.* **25,** 23430–23450.

Bellucci, C., Becker, M., Czarnowski, M. and Fitting, C. 2020. A novel method to evaluate stream connectivity using trail cameras. *River Research and Applications*. **36,** 1504-1514.

Bentley, K., Schindler, D., Armstrong, J., Zhang, R., Ruff, C. and Lisi, P. 2012. Foraging and growth responses of stream-dwelling fishes to inter-annual variation in a pulsed resource subsidy. *Ecosphere*. **3**, 113.

Birnie-Gauvin, K., Franklin, P., Wilkes, M. and Aarestrup, K. 2019. Moving beyond fitting fish into equations: Progressing the fish passage debate in the Anthropocene. *Aquatic conservation: Marine and freshwater ecosystems*. **29,** 1095-1105.

Black Sluice Internal Drainage Board. 2016. *Black Sluice Internal Drainage Board Website.* [Online]. [Accessed 14 January 2021]. Available from: https://www.blacksluiceidb.gov.uk

Bocker, E. 2015. *Restoring connectivity for migratory native fish: investigating the efficacy of Fish Friendly Gates.* MSc (Research) Thesis, Massey University, Palmerston North, New Zealand.

Bolland, J., Murphy, L., Stanford, R., Angelopoulos, N., Baker, N., Wright, R., Reeds, J. and Cowx, I. 2019. Direct and indirect impacts of pumping station operation on downstream migration of critically endangered European eel. *Fisheries Management and Ecology.* **26,** 76-85.

Boubée, J., Jowett, I., Nichols, S. and Williams, E. 1999. *Fish passage at culverts: a review, with possible solutions for New Zealand indigenous species.* Wellington, New Zealand: Department of Conservation.

Boys, C., Kroon, F., Glasby, T. and Wilkinson, K., 2012. Improved fish and crustacean passage in tidal creeks following floodgate remediation. *Journal of Applied Ecology*. **49,** 223-233.

Bradley, D., Cadman, D. and Milner, N. 2012. *Ecological indicators of the effects of abstraction and flow regulation; and optimisation of flow releases from water storage reservoirs*. Edinburgh, UK: Sniffer.

Branco, P., Segurado, P., Santos, J., Pinheiro, P. and Ferreira, M. 2012. Does longitudinal connectivity loss affect the distribution of freshwater fish?. *Ecological Engineering*. **48,** 70-78.

Brevé, N., Buijse, A., Kroes, M., Wanningen, H., and Vriese, F. 2014. Supporting decision-making for improving longitudinal connectivity for diadromous and potamodromous fishes in complex catchments. *Science of the Total Environment.* **496**, 206–218.

Briggs, A. and Galarowicz, T. 2013. Fish passage through culverts in central Michigan warmwater streams. *North American Journal of Fisheries Management.* **33,** 652-664.

Buddendorf, W., Jackson, F., Malcolm, I., Millidine, K., Geris, J., Wilkinson, M. and Soulsby, C. 2019. Integration of juvenile habitat quality and river connectivity models to understand and prioritise the management of barriers for Atlantic salmon populations across spatial scales. *Science of the Total Environment.* **655,** 557-566.

Cedarholm, C., Kunze, M., Murota, T. and Sibatani, A. 1999. Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. *Fisheries.* **24**, 6–15.

Childress, E. and McIntyre, P. 2015. Multiple nutrient subsidy pathways from a spawning migration of iteroparous fish. *Freshwater Biology*. **60,** 490-499.

Clapcott, J. 2015. *National rapid habitat assessment protocol development for streams and rivers (No. 2649).* Prepared for Northland Regional Council: Cawthron Institute. [Accessed 1 December 2019]. Available from:

https://www.cawthron.org.nz/media/publications/pdf/2015_03/CR2649_NATIONAL_RAPID_HABITAT_ASSESSMENT_PROTOCOL_DEVELOPMENT_FOR_STREAMS_AND_RIVERS.pdf

Collen, B., Whitton, F., Dyer, E., Baillie, J., Cumberlidge, N., Darwall, W., Pollock, C., Richman, N., Soulsby, A. and Böhm, M. 2014. Global patterns of freshwater species diversity, threat and endemism. *Global ecology and Biogeography*. **23,** 40-51.

Collevatti, R., Vitorino, L., Vieira, T., Oprea, M. and Telles, M. 2020. Landscape changes decrease genetic diversity in the Pallas' long-tongued bat. *Perspectives in Ecology and Conservation*. **18,** 169-177.

Cooney, P. and Kwak, T. 2013. Spatial extent and dynamics of dam impacts on tropical island freshwater fish assemblages. *BioScience*. **63,** 176-190.

Cote, D., Kehler, D., Bourne, C. and Wiersma, Y. 2009. A new measure of longitudinal connectivity for stream networks. *Landscape Ecology*. **24**, 101–113.

Crooks, K., Burdett, C., Theobald, D., King, S., Di Marco, M., Rondinini, C. and Boitani, L. 2017. Quantification of habitat fragmentation reveals extinction risk in terrestrial mammals. *Proceedings of the National Academy of Sciences*. **114,** 7635-7640.

Cumming, G. 2004. The impact of low-head dams on fish species richness in Wisconsin, USA. *Ecological Applications*. **14**, 1495–1506.

Dauble, D., Hanrahan, T., Geist, D. and Parsley, M. 2003. Impacts of the Columbia River hydroelectric system on main-stem habitats of fall Chinook salmon. *North American Journal of Fisheries Management*. **23,** 641-659.

David, B., Tonkin, J., Taipeti, K. and Hokianga, H. 2014. Learning the ropes: mussel spat ropes improve fish and shrimp passage through culverts. *Journal of applied ecology.* **51,** 214-223.

Dean, T. and Richardson, J. 1999. Responses of seven species of native freshwater fish and a shrimp to low levels of dissolved oxygen. *New Zealand Journal of Marine and Freshwater Research*. **33,** 99-106.

Denton, K., Rich, H., Moore, J. and Quinn, T. 2010. The utilization of a Pacific salmon *Oncorhynchus nerka* subsidy by three populations of charr *Salvelinus* spp. *Journal of Fish Biology*. **77**, 1006–1023.

Department of Marine Resources. 2020. *Trap Count Statistics.* [Online]. [Accessed 11 October 2020]. Available from: https://www.maine.gov

Dias, M., Cornu, J., Oberdorff, T., Lasso, C. and Tedesco, P. 2013. Natural fragmentation in river networks as a driver of speciation for freshwater fishes. *Ecography*. **36,** 683-689.

Didham, R., Tylianakis, J., Gemmell, N., Rand, T. and Ewers, R. 2007. Interactive effects of habitat modification and species invasion on native species decline. *Trends in ecology & evolution*. **22,** 489-496.

Doehring, K., Young, R. and McIntosh, A. 2011a. Factors affecting juvenile galaxiid fish passage at culverts. *Marine and Freshwater Research*. **62,** 38-45.

Doehring, K., Young, R. and McIntosh, A. 2012. Facilitation of upstream passage for juveniles of a weakly swimming migratory galaxiid. *New Zealand Journal of Marine and Freshwater Research.* **46,** 303-313.

Doehring, K., Young, R., Hay, J. and Quarterman, A. 2011b. Suitability of Dual-frequency Identification Sonar (DIDSON) to monitor juvenile fish movement at floodgates. *New Zealand Journal of Marine and Freshwater Research.* **45,** 413-422.

Dowling, L. 1999. Farming in Hawke's Bay. In *PROCEEDINGS OF THE CONFERENCE-NEW ZEALAND GRASSLAND ASSOCIATION*. **61,** 233-239.

Duncan, J. and Lockwood, J. 2001. Extinction in a field of bullets: a search for causes in the decline of the world's freshwater fishes. *Biological conservation.* **102,** 97-105.

Dunn, N., Allibone, R., Closs, G., Crow, S., David, B., Goodman, J., Griffiths, M., Jack, D., Ling, N., Waters, J. and Rolfe, J. 2018. *Conservation status of New Zealand freshwater fishes, 2017.* Wellington: Department of Conversation.

Environment Agency, 2018. *River Obstructions England and Wales*. Bristol: Environment Agency.

Fake, D. 2018. *Remediating small-scale migratory fish barriers with floating fish ramps.* MSc (Research) Thesis, The University of Waikato, Hamilton, New Zealand.

Ferguson, A., Reed, T., Cross, T., McGinnity, P. and Prodöhl, P. 2019. Anadromy, potamodromy and residency in brown trout *Salmo trutta*: the role of genes and the environment. *Journal of Fish Biology*. **95,** 692-718.

Fischer, J. and Lindenmayer, D. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global ecology and biogeography*. **16,** 265-280.

Fortune, J. and Mauraud, N. 2015. *Effect of tide on water quality of Jones Creek, Darwin Harbour.* Palmerston, Australia: Department of Land Resource Management.

Franklin, P. 2014. *National Fish Passage Management Symposium.* Hamilton, New Zealand: National Institute of Water and Atmospheric Research Ltd.

Franklin, P. and Bartels, B. 2012. Restoring connectivity for migratory native fish in a New Zealand stream: effectiveness of retrofitting a pipe culvert. *Aquatic Conservation: Marine and Freshwater Ecosystems*. **22,** 489-497.

Franklin, P. and Gee, E. 2019. Living in an amphidromous world: Perspectives on the management of fish passage from an island nation. *Aquatic Conservation: Marine and Freshwater Ecosystems*. **29,** 1424-1437.

Franklin, P. and Hodges, M. 2015. Modified tide gate management for enhancing instream habitat for native fish upstream of the saline limit. *Ecological Engineering. 81,* 233-242.

Fuller, M., Doyle, M. and Strayer, D. 2015. Causes and consequences of habitat fragmentation in river networks. *Annals of the New York Academy of Sciences. 1355*, 31–51.

Gao, X., Zeng, Y., Wang, J. and Liu, H. 2010. Immediate impacts of the second impoundment on fish communities in the Three Gorges Reservoir. *Environmental Biology of Fishes. 87,* 163-173.

Gehrke, P., Gilligan, D. and Barwick, M. 2002. Changes in fish communities of the Shoalhaven River 20 years after construction of Tallowa Dam, Australia. *River Research and Applications. 18,* 265-286.

Geist, J., and Hawkins, S. 2016. Habitat recovery and restoration in aquatic ecosystems: Current progress and future challenges. *Aquatic Conservation: Marine and Freshwater Ecosystems*. **26**, 942–962.

Giannico, G. and Souder, J. 2004. *The effects of tide gates on estuarine habitats and migratory fish*. Oregon State University: Sea Grant Oregon.

Gibson, L., Wilman, E., and Laurance, W. 2017. "How Green Is 'Green' Energy?" *Trends in Ecology & Evolution*. **32,** 922-935.

Goerig, E., Castro-Santos, T. and Bergeron, N. 2016. Brook trout passage performance through culverts. *Canadian Journal of Fisheries and Aquatic Sciences*. **73,** 94-104.

Google Earth. 2021. *Google Maps.* [Online]. [Accessed 19 January 2021]. Available from: https://earth.google.com

Gordos, M., Nichols, S., Lay, C., Townsend, A., Grove, C., Walsh, S. and Copeland, C. 2007. Audit and remediation of fish passage barriers in coastal NSW. In *Proceedings of the 5th Australian stream management conference. Australian rivers: making a difference*. **1,** 21-25.

Haggerty, J. 2007. "I'm not a greenie but…": Environmentality, eco-populism and governance in New Zealand Experiences from the Southland whitebait fishery. *Journal of Rural Studies*. **23,** 222-237.

Haidvogl, G., Hoffmann, R. and Pont, D. 2015. Historical ecology of riverine fish in Europe. *Aquat. Sci.* **77,** 315–324.

Han, M., Fukushima, M., Kameyama, S., Fukushima, T. and Matsushita, B. 2008. How do dams affect freshwater fish distributions in Japan? Statistical analysis of native and non-native species with various life histories. *Ecological Research*. **23,** 735-743.

Haro, A., Castro-Santos, T., Noreika, J. and Odeh, M. 2004. Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. *Canadian Journal of Fisheries and Aquatic Sciences.* **61**, 1591–1601.

Hawke's Bay Regional Council (HBRC). 2010. *Flood Control and Drainage Schemes.* [Online]. [Accessed 27 January 2020]. Available from: https://www.hbrc.govt.nz/services/flood-control/overview/

He, F., Zarfl, C., Bremerich, V., David, J., Hogan, Z., Kalinkat, G., Tockner, K. and Jähnig, S. 2019. The global decline of freshwater megafauna. *Global Change Biology*. **25,** 3883-3892.

Hermoso, V., Clavero, M., Blanco-Garrido, F. and Prenda, J. 2011. Invasive species and habitat degradation in Iberian streams: an analysis of their role in freshwater fish diversity loss. *Ecological Applications*. **21,** 175-188.

Hicks, A., Barbee, N., Swearer, S. and Downes, B. 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.

House, M., Pyles, M. and White, D. 2005. VELOCITY DISTRIBUTIONS IN STREAMBED SIMULATION CULVERTS USED FOR FISH PASSAGE 1. *Jawra Journal of the American Water Resources Association*. **41,** 209-217.

Investigate West (INVW). 2020. *Legislature, Inslee Struggle to Fix Roads That Block Salmon, Help Starve Orcas.* [Online]. [Accessed 14 January 2021]. Available from: https://www.invw.org

Januchowski-Hartley, S., McIntyre, P., Diebel, M., Doran, P., Infante, D., Joseph, C. and Allan, J. 2013. Restoring aquatic ecosystem connectivity requires expanding inventories of both dams and road crossings. *Frontiers in Ecology and the Environment*. **11,** 211-217.

Jellyman, D. and Graynoth, E. 2005. The use of fyke nets as a quantitative capture technique for freshwater eels (*Anguilla* spp.) in rivers. *Fisheries Management and Ecology*. **12,** 237-247.

Jellyman, P. and Harding, J. 2012. The role of dams in altering freshwater fish communities in New Zealand. *New Zealand Journal of Marine and Freshwater Research*. **46,** 475-489.

Johnson, P., Olden, J. and Vander Zanden, M. 2008. Dam invaders: impoundments facilitate biological invasions into freshwaters. *Frontiers in Ecology and the Environment*. **6,** 357-363.

Jones, J., Börger, L., Tummers, J., Jones, P., Lucas, M., Kerr, J., Kemp, P., Bizzi, S., Consuegra, S., Marcello, L. and Vowles, A. 2019. A comprehensive assessment of stream fragmentation in Great Britain. *Science of the total environment*. **673,** 756-762.

Joy, M. 2015. *Squandered: The degradation of New Zealand's freshwaters*. [Accessed 23 October 2021]. Available from: https://waterqualitynz.info/

Joy, M. and Death, R. 2001. Control of freshwater fish and crayfish community structure in Taranaki, New Zealand: dams, diadromy or habitat structure? *Freshwater Biology.* **46,** 417-429.

Katano, O., Nakamura, T., Abe, S., Yamamoto, S. and Baba, Y. 2006. Comparison of fish communities between above-and below-dam sections of small streams; barrier effect to diadromous fishes. *Journal of Fish Biology*. **68,** 767-782.

Kemp, P. and O'Hanley, J. 2010. Procedures for evaluating and prioritising the removal of fish passage barriers: a synthesis. *Fisheries Management and Ecology*. **17,** 297-322.

King, R., Stockley, B. and Stevens, J. 2020. Small coastal streams—Critical reservoirs of genetic diversity for trout *(Salmo trutta L.)* in the face of increasing anthropogenic stressors. *Ecology and Evolution*. **10,** 5651– 5669.

Kovach, R., Ellison, S., Pyare, S. and Tallmon, D. 2015. Temporal patterns in adult salmon migration timing across southeast Alaska. *Global change biology*. **21,** 1821-1833.

Krieg, R. and Zenker, A., 2020. A review of the use of physical barriers to stop the spread of non-indigenous crayfish species. *Reviews in Fish Biology and Fisheries.* **1,** 11-21.

Kroon, F. and Ansell, D. 2006. A comparison of species assemblages between drainage systems with and without floodgates: implications for coastal floodplain management. *Canadian Journal of Fisheries and Aquatic Sciences*. **63,** 2400-2417.

Lake, M. 2013. *Freshwater fish: passive nets-fyke nets.* [Online]. New Zealand: Department of Conservation Te Papa Atawhai. [Accessed 6 January 2022]. Available from: https://www.doc.govt.nz/globalassets/documents/science-and-technical/inventory-monitoring/im-toolbox-freshwater-fish/im-toolbox-freshwater-fish-passive-nets-fyke-nets.pdf

Land Information New Zealand (LINZ). 2020. *Tide Predictions.* [Online]. [Accessed 19 January 2021]. Available from: https://www.linz.govt.nz/sea/tides/tide-predictions

Lassalle, G., Béguer, M., Beaulaton, L. and Rochard, E. 2008. Diadromous fish conservation plans need to consider global warming issues: An approach using biogeographical models. *Biological conservation*. **141,** 1105-1118.

Lenders, H., Chamuleau, T., Hendriks, A., Lauwerier, R., Leuven, R. and Verberk, W. 2016. Historical rise of waterpower initiated the collapse of salmon stocks. *Sci. Rep.* **6,** 29269.

Levy, E., Byrne, M., Huey, J., Hillyer, M., Firman, R. and Ottewell, K. 2019. Limited influence of landscape on the genetic structure of three small mammals in a heterogeneous arid environment. *Journal of Biogeography*. **46,** 539-551.

Liermann, C., Nilsson, C., Robertson, J. and Ng, R. 2012. Implications of dam obstruction for global freshwater fish diversity. *BioScience*. **62,** 539-548.

Limburg, K. and Waldman, J. 2009. Dramatic declines in North Atlantic diadromous fishes. *BioScience*. **59,** 955-965.

Louca, V., Ream, H., Findlay, J., Latham, D. and Lucas, M. 2014. Do culverts impact the movements of the endangered white-clawed crayfish?. *Knowledge and Management of Aquatic Ecosystems*. **414,** 14.

Lucas, M. and Frear, P. 1997. Effects of a flow-gauging weir on the migratory behaviour of adult barbel, a riverine cyprinid. *Journal of fish biology*. **50,** 382-396.

Lynch, A., Cooke, S., Deines, A., Bower, S., Bunnell, D., Cowx, I., Nguyen, V., Nohner, J., Phouthavong, K., Riley, B. and Rogers, M. 2016. The social, economic, and environmental importance of inland fish and fisheries. *Environmental Reviews*. **24,** 115-121.

MacDonald, J. and Davies, P. 2007. Improving the upstream passage of two galaxiid fish species through a pipe culvert. *Fisheries Management and Ecology.* **14,** 221–230.

Mallen-Cooper, M. and Brand, D. 2007. Non-salmonids in a salmonid fishway: what do 50 years of data tell us about past and future fish passage?. *Fisheries Management and Ecology*. **14,** 319-332.

McCarthy, T., Frankiewicz, P., Cullen, P., Blaszkowski, M., O'Connor, W. and Doherty, D. 2008. Long-term effects of hydropower installations and associated river regulation on River Shannon eel populations: mitigation and management. *Hydrobiologia*. **609,** 109-124.

McDowall, R. 1990. When galaxiid and salmonid fishes meet—a family reunion in New Zealand. *Journal of Fish Biology*. **37,** 35-43.

McDowall, R. 1996. Volcanism and freshwater fish biogeography in the north-eastern North Island of New Zealand. *Journal of biogeography*. **23,** 139-148.

McIntosh, A., McHugh, P., Dunn, N., Goodman, J., Howard, S., Jellyman, P., O'Brien, L., Nyström, P. and Woodford, D. 2010. The impact of trout on galaxiid fishes in New Zealand. *New Zealand Journal of Ecology.* **34,** 195-206.

McIntyre, P., Jones, L., Flecker, A. and Vanni, M. 2007. Fish extinctions alter nutrient recycling in tropical freshwaters. *Proceedings of the National Academy of Sciences*. **104,** 4461-4466.

McLaughlin, R., Smyth, E., Castro-Santos, T., Jones, M., Koops, M., Pratt, T. and Vélez-Espino, L. 2013. Unintended consequences and trade-offs of fish passage. *Fish and Fisheries*. **14,** 580-604.

McRae, N., Gaw, S. and Glover, C. 2016. Mechanisms of zinc toxicity in the galaxiid fish, *Galaxias maculatus*. *Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology*. **179,** 184-190.

McRae, N., Gaw, S. and Glover, C. 2018. Effects of waterborne cadmium on metabolic rate, oxidative stress, and ion regulation in the freshwater fish, inanga *(Galaxias maculatus). Aquatic Toxicology*. **194,** 1-9.

Meixler, M., Bain, M. and Walter, M. 2009. Predicting barrier passage and habitat suitability for migratory fish species. *Ecological Modelling*. **220**, 2782-2791.

Merckx, T., Souffreau, C., Kaiser, A., Baardsen, L., Backeljau, T., Bonte, D., Brans, K., Cours, M., Dahirel, M., Debortoli, N. and De Wolf, K. 2018. Body-size shifts in aquatic and terrestrial urban communities. *Nature*. **558,** 113-116.

Miller, R., Bradford, W. and Peters, N. 1988. *Specific conductance: theoretical considerations and application to analytical quality control,* Vol. 142. Washington, DC: US Government Printing Office.

Miserendino, M., Casaux, R., Archangelsky, M., Di Prinzio, C., Brand, C. and Kutschker, A. 2011. Assessing land-use effects on water quality, in-stream habitat, riparian ecosystems and biodiversity in Patagonian northwest streams. *Science of the total environment*. **409,** 612-624.

Mitchell, C. 1989. Swimming performances of some native freshwater fishes. *New Zealand Journal of Marine and Freshwater Research. ***23,** 181–187.

Mueller, M., Pander, J. and Geist, J. 2011. The effects of weirs on structural stream habitat and biological communities. *Journal of Applied Ecology. ***48**, 1450–1461.

Nelson, B., Sasekumar, A. and Ibrahim, Z. 1994. Neap-spring tidal effects on dissolved oxygen in two Malaysian estuaries. *Hydrobiologia. ***285,** 7-17.

New Zealand Bio-Recording Network Trust. 2021. *iNaturalistNZ.* [Online]. [Accessed 15 July 2021]. Available from: https://inaturalist.nz/

Ngāi Tahu Claims Settlement Act 1998. Wellington: New Zealand Government.

Nieminen, E., Hyytiäinen, K. and Lindroos, M. 2017. Economic and policy considerations regarding hydropower and migratory fish. *Fish and Fisheries*. **18,** 54-78.

Nislow, K., Hudy, M., Letcher, B. and Smith, E. 2011. Variation in local abundance and species richness of stream fishes in relation to dispersal barriers: implications for management and conservation. *Freshwater Biology. ***56,** 2135-2144.

Obolewski, K., Glińska-Lewczuk, K. and Astel, A. 2018. Lost connectivity between a coastal lagoon and the sea–implications of floodgate closure for benthic macroinvertebrates. *Estuarine, Coastal and Shelf Science*. **211,** 77-89.

O'Hara, R. and Kotze, J. 2010. Do not log-transform count data. *Nature Precedings*. **4136,** 1.

Orchard, S. and Hickford, M. 2016. *Spatial effects of the Canterbury earthquakes on inanga spawning habitat and implications for waterways management.* [Online]. Christchurch: University of Canterbury. [Accessed 16 December 2020]. Available from: https://ir.canterbury.ac.nz/bitstream/handle/10092/13524/

Ovidio, M., Capra, H. and Philippart, J. 2007. Field protocol for assessing small obstacles to migration of brown trout *Salmo trutta*, and European grayling *Thymallus thymallus*: a contribution to the management of free movement in rivers. *Fisheries Management and Ecology.* **14**, 41–50.

Park, D., Sullivan, M., Bayne, E. and Scrimgeour, G. 2008. Landscape-level stream fragmentation caused by hanging culverts along roads in Alberta's boreal forest. *Canadian Journal of Forest Research*. **38,** 566-575.

Pelicice, F., Pompeu, P. and Agostinho, A. 2015. Large reservoirs as ecological barriers to downstream movements of Neotropical migratory fish. *Fish and Fisheries*. **16,** 697-715.

Piccolo, J., Norrgård, J., Greenberg, L., Schmitz, M. and Bergman, E. 2012. Conservation of endemic landlocked salmonids in regulated rivers: a case-study from Lake Vänern, Sweden. *Fish and Fisheries*. **13,** 418-433.

Poff, N. 2014. Rivers of the Anthropocene? *Frontiers in Ecology and the Environment*. **12,** 427-427.

Poff, N. and Schmidt, J. 2016. How dams can go with the flow. *Science*. **353,** 1099-1100.

Pollard, D. and Hannan, J. 1994. The ecological effects of structural flood mitigation works on fish habitats and fish communities in the lower Clarence River system of south-eastern Australia. *Estuaries*. **17,** 427-461.

Portt, C., Coker, G., Ming, D. and Randall, R. 2006. A review of fish sampling methods commonly used in Canadian freshwater habitats. *Canadian Technical Report of Fisheries and Aquatic Sciences.* **2604,** 1-48.

Poulin, R., Closs, G., Lill, A., Hicks, A., Herrmann, K. and Kelly, D. 2012. Migration as an escape from parasitism in New Zealand galaxiid fishes. *Oecologia*. **169,** 955-963.

Radinger, J. and Wolter, C. 2014. Patterns and predictors of fish dispersal in rivers. *Fish and fisheries*. **15,** 456-473.

Richardson, J. and Taylor, M. 2002. *A guide to restoring inanga habitat.* Wellington, NZ: NIWA.

Richardson, J., Boubée, J. and West, D. 1994. Thermal tolerance and preference of some native New Zealand freshwater fish. *New Zealand journal of marine and freshwater research*. **28,** 399-407.

Richey, J. and Swanson, F. 1989. The river continuum concept: a basis for the expected ecosystem behaviour of very large rivers?. *Canadian journal of fisheries and aquatic sciences*. **106,** 49-55.

Richter, B., Braun, D., Mendelson, M. and Master, L. 1997. Threats to imperilled freshwater fauna. *Conservation Biology*. **11,** 1081-1093.

Rincón, G., Solana-Gutiérrez, J., Alonso, C., Saura, S., and García de Jalón, D. 2017. Longitudinal connectivity loss in a riverine network: Accounting for the likelihood of upstream and downstream movement across dams. *Aquatic Sciences*. **79**, 573–585.

Rivers-Moore, N., Mantel, S., Ramulifo, P. and Dallas, H. 2016. A dysconnectivity index for improving choices in managing protected areas for rivers. *Aquatic Conservation: Marine and Freshwater Ecosystems*. **26,** 29-38.

Roberts, J., Angermeier, P. and Hallerman, E. 2013. Distance, dams and drift: what structures populations of an endangered, benthic stream fish?. *Freshwater Biology*. **58,** 2050-2064.

Rodgers, E., Heaslip, B., Cramp, R., Riches, M., Gordos, M. and Franklin, C. 2017. Substrate roughening improves swimming performance in two small-bodied riverine fishes: implications for culvert remediation and design. *Conservation physiology*. **5,** 1-10.

Rolls, R. 2011. The role of life-history and location of barriers to migration in the spatial distribution and conservation of fish assemblages in a coastal river system. *Biological conservation*. **144,** 339-349.

Roni, P., Beechie, T., Bilby, R., Leonetti, F., Pollock, M. and Pess, G. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management.* **22**, 1-20.

Roni, P., Hanson, K. and Beechie, T. 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. *North American Journal of Fisheries Management.* **28**, 856-890.

Roy, A., Rosemond, A., Paul, M., Leigh, D. and Wallace, J. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). *Freshwater biology*. **48,** 329-346.

Russell, I., Moore, A., Ives, S., Kell, L., Ives, M. and Stonehewer, R. 1998. The migratory behaviour of juvenile and adult salmonids in relation to an estuarine barrage. *Hydrobiologia*. **371,** 321-334.

Schick, R. and Lindley, S. 2007. Directed connectivity among fish populations in a riverine network. *Journal of Applied Ecology.* **44**, 1116–1126.

Schinegger, R., Trautwein, C., Melcher, A. and Schmutz, S. 2012. Multiple human pressures and their spatial patterns in European running waters. *Water and Environment Journal*. **26,** 261-273.

Segurado, P., Branco, P., Avelar, A., Ferreira, M. 2015. Historical changes in the functional connectivity of rivers based on spatial network analysis and the past occurrences of diadromous species in Portugal. *Aquat. Sci.* **77,** 427–440.

Sheer, M. and Steel, E. 2006. Lost watersheds: barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and Lower Columbia River basins. *Transactions of the American Fisheries Society*. **135**, 1654–1669.

Silva, A., Lucas, M., Castro-Santos, T., Katopodis, C., Baumgartner, L., Thiem, J., Cooke, S. 2018. The future of fish passage science, engineering and Practice. *Fish and Fisheries*. **19**, 340–362.

Stevens, J., Hickford, M. and Schiel, D. 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.

Stevens, J., Kocik, J. and Sheehan, T. 2019. Modelling the impacts of dams and stocking practices on an endangered Atlantic salmon *(Salmo salar)* population in the Penobscot River, Maine, USA. *Canadian Journal of Fisheries and Aquatic Sciences*. **76,** 1795-1807.

Van Puijenbroek, P., Buijse, A., Kraak, M. and Verdonschot, P. 2019. Species and river specific effects of river fragmentation on European anadromous fish species. *River Res. Appl.* **35,** 68–77.

Vannote, R., Minshall, G., Cummins, K., Sedell, J. and Cushing, C. 1980. The river continuum concept. *Canadian journal of fisheries and aquatic sciences*. **37,** 130-137.

Vörösmarty, C., McIntyre, P., Gessner, M., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S., Sullivan, C. and Liermann, C. 2010. Global threats to human water security and river biodiversity. *Nature*. **467,** 555.

Warton, D. 2005. Many zeros does not mean zero inflation: comparing the goodness-of-fit of parametric models to multivariate abundance data. *Environmetrics: The official journal of the International Environmetrics Society*. **16,** 275-289.

Wild Trout Trust. 2019. *Improvise, adapt, and overcome*. [Online]. [Accessed 14 January 2021]. Available from: https://www.wildtrout.org

Williams, J., Armstrong, G., Katopodis, C., Larinier, M. and Travade, F. 2012. Thinking like a fish: a key ingredient for development of effective fish passage facilities at river obstructions. *River Research and Applications*. **28,** 407-417.

Willson, M. and Halupka, K. 1995. Anadromous fish as keystone species in vertebrate communities. *Conservation Biology*. **9**, 489–497.

Winter, H. and Van Densen, W. 2001. Assessing the opportunities for upstream migration of non-salmonid fishes in the weir-regulated River Vecht. *Fisheries Management and Ecology*. **8,** 513-532.

Wofford, J., Gresswel, R. and Banks, M. 2005. Influence of barriers to movement on within-watershed genetic variation of coastal cutthroat trout. *Ecological Applications.* **15**, 628–637.

Wohl, E. 2005. Compromised rivers: understanding historical human impacts on rivers in the context of restoration. *Ecology and Society*. **10,** 1-16.

Wohl, E. 2017. Connectivity in rivers. *Progress in Physical Geography*. **41,** 345-362.

Wright, G., Wright, R., Bendall, B. and Kemp, P. 2016. Impact of tide gates on the upstream movement of adult brown trout, *Salmo trutta*. *Ecological Engineering*. **91,** 495-505.

Yungnickel, M. 2017. *New Zealand's whitebait fishery: spatial and temporal variation in species composition and morphology.* MSc Thesis, University of Canterbury, New Zealand.