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Incorporating fish monitoring into the Waikato Regional Councils' Regional Ecological Monitoring of Streams (REMS) – preliminary results for wadeable streams 2009-2015.

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Executive Summary

In this report we review preliminary data from the Waikato Regional Council's State of the Environment (SOE) wadeable rivers fishing programme. Standardized fisheries sampling commenced in 2009 at a number of least impaired reference sites and by 2011/2012 routine fisheries sampling was added to the Councils existing Regional Ecological Monitoring of Streams (REMS) programme. Due to the migratory nature of many native fish species, these organisms can provide useful information on riverine connectivity between freshwater environments and the ocean. In this report we review data collected from annually monitored fish reference sites (since 2009) and also data from the first 3 year rotation of sites from the random probabilistic wadeable river network. In exploring the data we firstly focus on evaluating the consistency of specific protocol elements over time at our longest monitored reference sites. In short data suggested low variability between years for a range of measured parameters such as relative abundance and diversity of fish and physical and chemical parameters at individual reference sites. We then evaluated both the reference and random network and discuss the potential for fish metrics such as the Fish Index of Biological Integrity (Fish IBI) to be used for reporting on river condition. We also illustrate the utility of the randomized probabilistic network for reporting on extent estimates of fish metrics by using the FishIBI as an example across the Waikato wadeable river network. As a further component of this report we examine some species specific patterns that appear to be emerging across the region for frequently encountered species like long and shortfin eels and redfin bullies. In brief over this six year period of assessment longfin elvers (<100mm) were rarely captured throughout the network compared to shortfin elvers and there is a tendency for shortfin elvers to be detected at sites that are predominantly or exclusively dominated by larger size classes (>100mm) of longfin eels. Redfin bullies on the other hand appeared to be on average smaller but exhibit greater recruitment potential in streams draining to the east compared to sites draining to the west of the region. In the future it is envisaged that refined fish metrics could be combined with other physical, chemical and ecological components (including invertebrates) to better describe and report on the 'health' of individual wadeable river reaches.

1 Introduction

Ecologically based State of Environment (SOE) river assessments in New Zealand have historically relied on aquatic plants (including algae) and macroinvertebrates for assessing 'health' along with other traditional water quality parameters (e.g. nutrients, *E. coli* etc). Since river networks form a wetted connection from source to sea, a range of 'non-winged' aquatic biota are often reliant on unimpaired connectivity for exploiting feeding opportunities and to complete their life-cycles. Many barriers of varying scale exist on river networks can disrupt riverscape fish diversity (Jellyman & Harding 2012). Therefore, an important consideration for any riverine health assessment is also an understanding of the degree of connectedness of the sampled reach within the system. In New Zealand, numerous freshwater fish species and one crustacean, the shrimp *Paratya curvirostris,* move regularly throughout river networks and around one third of them are diadromous. For many widely distributed diadromous species like the endemic longfin eel (*Anguilla dieffenbachii*), bluegill bully (*Gobiomorphus hubbsi*) and torrentfish (*Cheimarrichthys fosteri*) this life-history strategy appears to be obligatory. Thus incorporation of fish community 'metrics' into traditional reach scale river health assessments can provide additional important data to inform the integrity of riverscape connectivity and river condition.

Since approximately one third of New Zealand's native freshwater fish species use the ocean as part of their lifecycle, many of the same species can be found throughout both the North and South Islands particularly in catchments close to the coast. Because of this life-history strategy, juveniles born in one region may disperse at sea and could potentially recruit to other regions. Thus to confidently establish the state of these stocks and to enable appropriate regional assessment and management, a consistent and coordinated approach to assessing their populations is necessary.

Notwithstanding their important recreational (e.g. trout, whitebait), customary and commercial value (e.g. eels, mullet), freshwater fish are a vital component of freshwater biodiversity. Consequently various fish assemblage attributes can provide unique and useful information in addition to other traditional measures (e.g. water quality) to assist in a more robust assessment of river condition and river health. For instance, because many fish in New Zealand are long lived (1-80+ yrs) and many are migratory, the presence or absence of particular fish at a particular place can provide information on both present and past conditions at a site.

For the past 8 years the Waikato Regional Council in association with other regional councils and Massey University has been pioneering the development and implementation of robust and repeatable standardised protocols for sampling fish communities in wadeable New Zealand streams. The primary aim of these protocols has been to evaluate a selection of methods to effectively and robustly report on the reach scale diversity and relative abundance of fish communities across a diverse range of New Zealand wadeable streams (David et al. 2010, David & Hamer 2010, Joy et al. 2013).

In addition to protocol development, the Waikato Regional Council has also led the development of an electronic field data capture system for each fish assessment method to improve data capture efficiencies, reduce transcription errors and ensure consistency of information collection and storage within and external to our region. The protocols and data capture system (which populates a standard excel based upload template) have been developed for regional and national use to improve data collection consistency and ultimately improve the ability to evaluate regional and national trends in wadeable riverine fish communities. At present the paucity of long-term and consistently collected fish community data has hampered the exploration and development of potentially useful fish metrics to assess the national state and trend of fish populations in New Zealand rivers. Although a platform now exists to collect this information for wadeable stream fish communities (excluding braided river beds), significant gaps still exist with respect to protocol development for robust evaluation of fish communities in lakes, wetlands and large non wadeable rivers in New Zealand. Consistent collection of such data at both regional and national scales is becoming increasingly important considering that currently more than two thirds of the New Zealand's native fish taxa are listed as 'declining' or worse (Goodman et al. 2014).

In this preliminary report we review fishing data from the first complete 3 year rotation of sites which form part of the existing REMS network for macroinvertebrate sampling. Data collected from annually monitored fish reference sites since 2009 is also reported. The potential for using this information for developing fish metrics for future combination with other physical, chemical and ecological components to better describe the 'health' of individual wadeable river reaches is also explored.

2 Methods

2.1 Fishing Methods

Protocol

Protocols for standardised sampling of fish communities in wadeable streams are described in detail in David & Hamer (2010) and Joy et al. (2013). Briefly, the spatial scale of assessment is 150 stream metres. This particular distance is sampled because previous protocol development across a wide range of wadeable New Zealand rivers (up to 12m wetted width) indicated that this is the distance at which fish species diversity tends to asymptote at a reach scale using one pass electrofishing (David et al. 2010). This work also indicated that distance rather than area fished was a better predictor of diversity and if distances up to 150m were sampled then there was a >90% likelihood of detecting the diversity of fish likely to be present at a reach scale. Although additional species could be detected beyond this distance in some circumstances, the additional survey effort required for species return would likely be cost and time prohibitive for little additional gain (David et al. 2010). Consequently all SOE fish sampling undertaken by Waikato Regional Council in wadeable streams (including netting protocols) are conducted at this spatial scale of assessment.

Fishing method selection

Upon visiting a site for the first time the standardised methodology decision support assessment table of Joy et al. (2013) was used to determine the most appropriate sampling methodology for that site, i.e. whether the site should be electrofished, netted or spotlighted. In this particular SOE programme all sites were either electrofished or netted. While spotlighting is a useful method, it is typically only advantageous over the other methods in specific situations and for specific species. Furthermore there is likely to be greater variance amongst observers using this methodology (based on their experience) than electrofishing or netting. Spotlighting was never assessed as the best method for a site, and for the above reasons, where spotlighting scored equal to one of the other preferred methodologies as the preferred method, we selected the alternative (i.e. netting or electrofishing. In those situations where electrofishing and netting scores were equal to each other, we used experience to decide which method to use. Once a method was selected for a site, the same method was applied for any future visits to ensure methodological comparability. For a very small number of random sites, sampling was either not possible in a particular year (e.g. dry due to drought) or a change in methodology from electrofishing to netting between rotations was enacted. Method change was mainly driven by atypically low water enabling electrofishing in the first sampling year but not under more typical summer base-flow conditions in subsequent years. Consequently a shift to netting at these sites was made to ensure a more usable long term methodology for these sites. Sites that were dry or that underwent a method change were excluded from analyses for this report but are identified in Table 1.

The fishing protocol employed at the majority of sites was single pass electrofishing. While all reference sites were surveyed using electrofishing, not all developed land sites were able to be surveyed using this method. A relatively small number of sites on developed land were not conducive to electrofishing and at these sites standardised trapping protocols (fyke and minnow traps also set over 150m) were used.

2.2 Network Development and Site Selection

Minimally impaired Reference sites for fish

An important requirement for assessing impacts on fish from anthropogenic effects is to sample a set of minimally impaired reference sites against which developed metrics can be compared. Various criteria can be used to define 'minimally impaired'. The REMS invertebrate sampling programme has an existing set of 'reference' sites but when the criteria were developed, riverscape connectivity (to the site) was not specifically identified as a key criteria (see Collier et al. 2007, Collier & Hamer 2010). The presence of artificial barriers on river networks may have little influence on local invertebrate communities since they can reproduce locally (excluding the diadromous and endemic shrimp *Paratya curvirostris*) but barriers can impede different fish species to different degrees. Unimpeded connectivity is a critical consideration for many of New Zealand's native fish species as more than one third exhibit diadromous life-histories, that is, they require access to and from the marine environment at some stage to complete their life-cycle. Consequently a desktop assessment was initially undertaken to identify potential minimally impaired reference sites for fish if they met the following criteria:

- Upstream reach land-use >70% indigenous vegetation cover
- No historical or active mines in catchment
- No artificial downstream or upstream barriers
- >90% natural flow regime (minimal abstraction pressure)
- >90% fish/invertebrate/algal assemblage composed of native species
- Wadeable = >90% stream bed <0.6m deep

In New Zealand it is well recognised that fish diversity decreases with distance inland and altitude. Additionally there is a growing body of evidence to suggest that coastal processes (currents, harbours etc.) may influence dispersal or retention of larvae and potentially recruitment of some species back to freshwater (Hicks 2012, Warburton 2016). In recognition of these possible effects we developed the following categories to generate a representative spread of regional reference sites:

Distance inland categories:

- <10, 10-30, 30 + km inland

Coastal morphology and location:

- East coast – flow path to open coast
- East coast harbour - flow path to inlet or harbour
- West coast open - flow path to open coast
- West coast harbour - flow path to inlet or harbour

An acknowledged shortcoming of the WRC reference site network at present is that we have been unable to find sufficient reference sites meeting our criteria for benchmarking expected fish communities in low elevation sites > 30km inland. Future reports will assess the potential to model and re-construct likely fish communities (e.g. Leathwick et al. 2005) that would have existed, possibly by applying historical vegetation layers or using other existing examples in other regions and developing an expected historical 'Fish Index of Biological Integrity'. Of the reference sites we did find, and that were suitable for sampling, they varied with respect to stream size (primarily 2-4th order) distance inland (1.6 - 80km) and altitude (15 - 187m asl, Fig. 1). To ensure comparability with the REMS programme, identical REMS invertebrate sampling and associated protocols were adopted at all fish reference sites that were sampled.

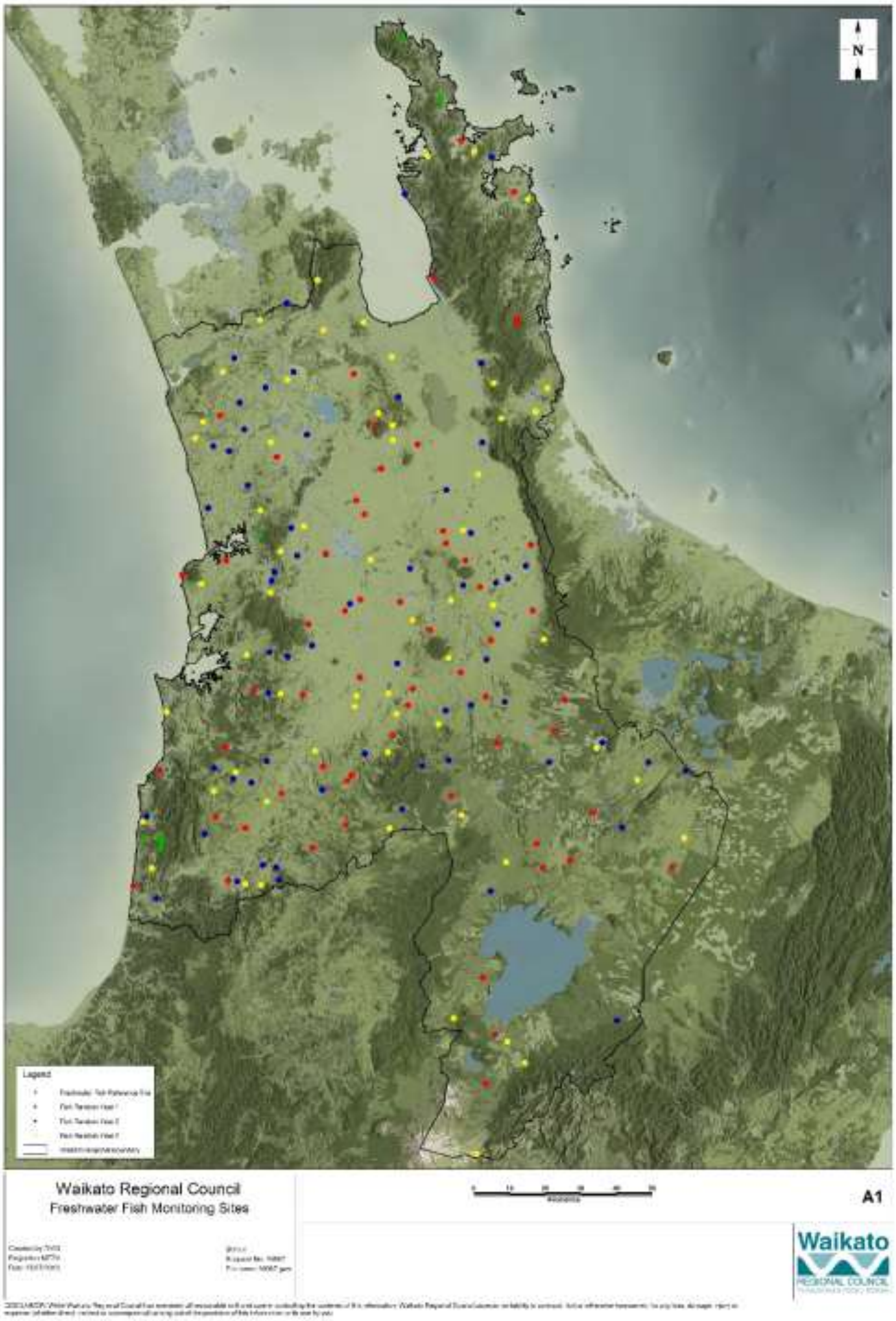


Figure 1: Location of all electrofishing sites in the Waikato regional Council fish monitoring programme. Green dots = reference sites, Red, blue and yellow dots = random year 1, 2 and 3 year sites respectively

Random sites – Fish

While it appears necessary to incorporate additional 'reference' sites specifically for fish into the REMS programme, the existing REMS random site network was used to assess fish communities on developed land across the Waikato region. The REMS random site network, currently sampled for macroinvertebrates, is made up of 180 sites. Sixty of these target sites are sampled each year on a rotating 3-yearly basis, this provides 180 samples from 180 different sites over a given three year period. The probability survey design was implemented by randomly selecting wadeable sites on developed land with known probability of inclusion using the survey design software package spsurvey :

(https://www.nemi.gov/methods/sams_method_summary/11950/).

The target population for site selection was non-reference (i.e. on developed land), non-tidal, perennial, wadeable streams. Equal numbers of 1st, 2nd, 3rd and ≥4th order streams were selected (i.e. balanced unequal probability design) using the REC river network layer as the sample frame (Collier & Hamer 2012, Fig 1.). For various reasons (e.g. permission to access denied, no flow due to drought) some sites were not able to be sampled within a given year. Also, a slight difference between 'wadeable' suitability criteria exists between the invertebrate and fishing programmes. In particular, a small number of some of the larger rivers that could be sampled using the invertebrate protocols were too wide and or deep for effective fish sampling using the standardised fish sampling protocols.

Parameters measured at random and reference sites

At all sampled sites a range of quantitative and qualitative physical, chemical and biological parameters were measured (Appendix 1). Parameters measured included information on physical habitat and condition, macroinvertebrates, periphyton, macrophytes, shade and water quality (including point in time water samples collected for later analyses of nutrients and faecal coliforms). Most of these other parameters will be combined and analysed as part of a separate report on combined river health metrics.

The fish community at each site was assessed by sampling a 150m reach, all other parameters (invertebrates, habitat, etc.) were evaluated from a 100m reach nested within each 150m fished stream reach (i.e. fishing was conducted over an additional 25m upstream and downstream of the existing REMS site). In most cases these parameters (excluding fish) were recorded and assessed as part of the existing REMS macroinvertebrate assessment, carried out typically 1-4 weeks prior to or following fish sampling (during stable conditions). In taking this approach we have made the assumption that parameters (other than fish) measured at a centralised 100m within the fish site would be biologically, physically and chemically representative to parameters had they been measured at a 150m scale. We have also assumed that any disturbance to the stream reach (caused by the invertebrate team while collecting samples and other parameters) would have completely recovered by the time fishing was undertaken or vice versa.

Seasonal timing of surveys

WRC's SOE fishing programme begins annually on December 1 and finishes by April 30th. This corresponds to the time of year when fish are most active and susceptible to capture. Irrespective of location and as described in WRC's standardised fishing protocols, all sites electrofished on more than one occasion were sampled at exactly the same GPS location using the same back pack EF machine and machine settings used previously. Every attempt was made to apply the same level of effort (button 'shock' time) at a site as previously. In some cases slight differences in amount of fishing time was unavoidable due to physical changes to the survey reach. These included instances where some sections may have been fishable on one occasion but were not fishable on subsequent occasions and vice versa (e.g. due to excessive depth or large debris preventing access to the channel). Another factor that may have influenced button 'shock' time was average channel wetted width and associated water velocity which influences fishing efficiency and may have been slightly more or less on any given visit (depending on stream flows). All parameters (except for machine settings and shock 'button' time) measured on each sampling occasion at a site were measured without knowledge of values generated in previous years to ensure independence and to minimize protocol bias. For electrofishing, the value for machine shock 'button' time (expressed as a range from minimum to maximum shock time at a site) was specifically recalled from previous sampling to ensure a similar effort was expended at the site.

To further minimise protocol variance, we also attempted to sample repeat sites at the same time of year (within 1-2 weeks) as previously. It is important to recognize that such an approach could introduce a seasonal bias between sites, particularly with respect to summer recruitment timing of some species into freshwater. Nevertheless we have assumed that the ability to detect any changes at a specific site over time will be greater with this type approach and that additional future information could be collected to investigate any seasonal bias that may exist. Occasionally adverse weather and bed-moving flood stand-down protocols were enacted forcing some sites to be sampled later. With respect to bed moving events we utilise a regional network of telemetered flow monitoring sites which have established bed moving flow trigger values (Appendix 2). If this value is triggered, a precautionary minimum stand-down period of two weeks is enacted for any sites proximal to the flow site recorder. Although we have limited data to support the value of a stand-down period to minimise variance for biological monitoring, we have assumed that significant bed moving flow events would alter the structure and function of local communities (in the short term) from that exhibited during typical base-flow conditions. Data for fish species in New Zealand rivers suggests that different fish may respond differentially to flood events with some species apparently affected and others less so or not at all (e.g. Hayes et al. 2010; McEwan & Joy 2013b).

Sites which were required to be netted tended to be slower flowing, slightly deeper and dominated by aquatic vegetation. These sites were generally sampled towards the end of the sampling season (late March/April). The decision to net later in the season within the Waikato region is primarily to minimise overnight mortalities that can result from high temperatures and low oxygen driven by aquatic plant respiration. Mortalities may be compounded by large numbers of fish accumulating and respiring themselves within the confined space of the nets. Although it is recognised that only netting late into season could also introduce a form of bias into the dataset, a trade-off between potential mortality of fish (which could also introduce a method impact bias at a site) and seasonal timing was necessary. As with electrofishing, for repeat netting sites the same GPS co-ordinates and seasonal timing were used to set nets in the same places and times if weather and site conditions permitted. Given the relatively low proportion of netted sites, data from this method are only briefly covered in the results.

Data analysis structure

For the purpose of this report analyses have been broken into three categories 'Reference' (sites sampled from annually 2009/10-2014/15), 'Random' (sites sampled first in 2011/12 and then repeat fished in 2014/15) and 'Reference and Random' (where only data from 2011/12 and 2014/15 sampling years were used). Additionally a preliminary assessment of the full set of sites (2011/12 – 2014/15) is undertaken to demonstrate the potential for reporting on regional extent estimates.

In this first 3yr rotational assessment of the fish programme preliminary analyses were undertaken to:

- Evaluate methodological consistency at sites between years
- Examine any temporal physical, chemical and biological patterns in the reference network
- Evaluate the potential for developing specific fish indicators for assessment of ecological health at both a site and regional scale

- Provide a regional summary of fish community attributes for the first set of repeat visit random sites (1st 3 yr rotation) and provide a summary and preliminary network extent estimates for the first full complete series of sites (2011/12 – 2014/15).

3 Results

Reference sites – assessment of methodological variance 2009/10 – 2014/15

Since the inception of WRC's fish sampling programme in 2009/2010, 15 reference sites have been sampled. Of these, 11 have been fished on three occasions or more (different years) with the longest running sites having been fished annually for the last 6 consecutive years (Table 1). Four sites have only been visited on a single occasion thus far but will likely be re-surveyed in the future.

Table 1: Summary of random and reference sites and number by fished sampling method from 2009/2010 to 2014/2015. The number of sites in brackets represents the number of sites in a given year which were surveyed using the netting protocol, all others were electrofished. [1] 2 sites were electrofished in 2011/12 but not in 2014/15 as were dry, and for one site, site access was denied in 2014/15

Fished Sites	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15
Random sites	0	4	42[1]	50 (7)	53 (14)	54 (12) [1]
Reference sites	6	10	10	10	11	11
Total	6	14	52	60	64	65

Physical and methodological parameters:

In general there was very little variability in the average wetted stream width or average total area fished over time at individual reference sites (Fig. 2 A, B). With respect to wetted area, variance is minimised by ensuring that sampling is conducted under stable (typically base flow) conditions and that wetted widths are sampled at the exact same permanently marked transect locations on each occasion. Consequently, any difference in the relative abundance of fish (fish/100m^2) at a site over time was likely to be minimally influenced by these two parameters. Stream area fished at reference sites ranged from just under 300m^2 (2nd order Stony Bay site) to just over 800m^2 (3rd order Stony Bay site) (Fig. 2 B).

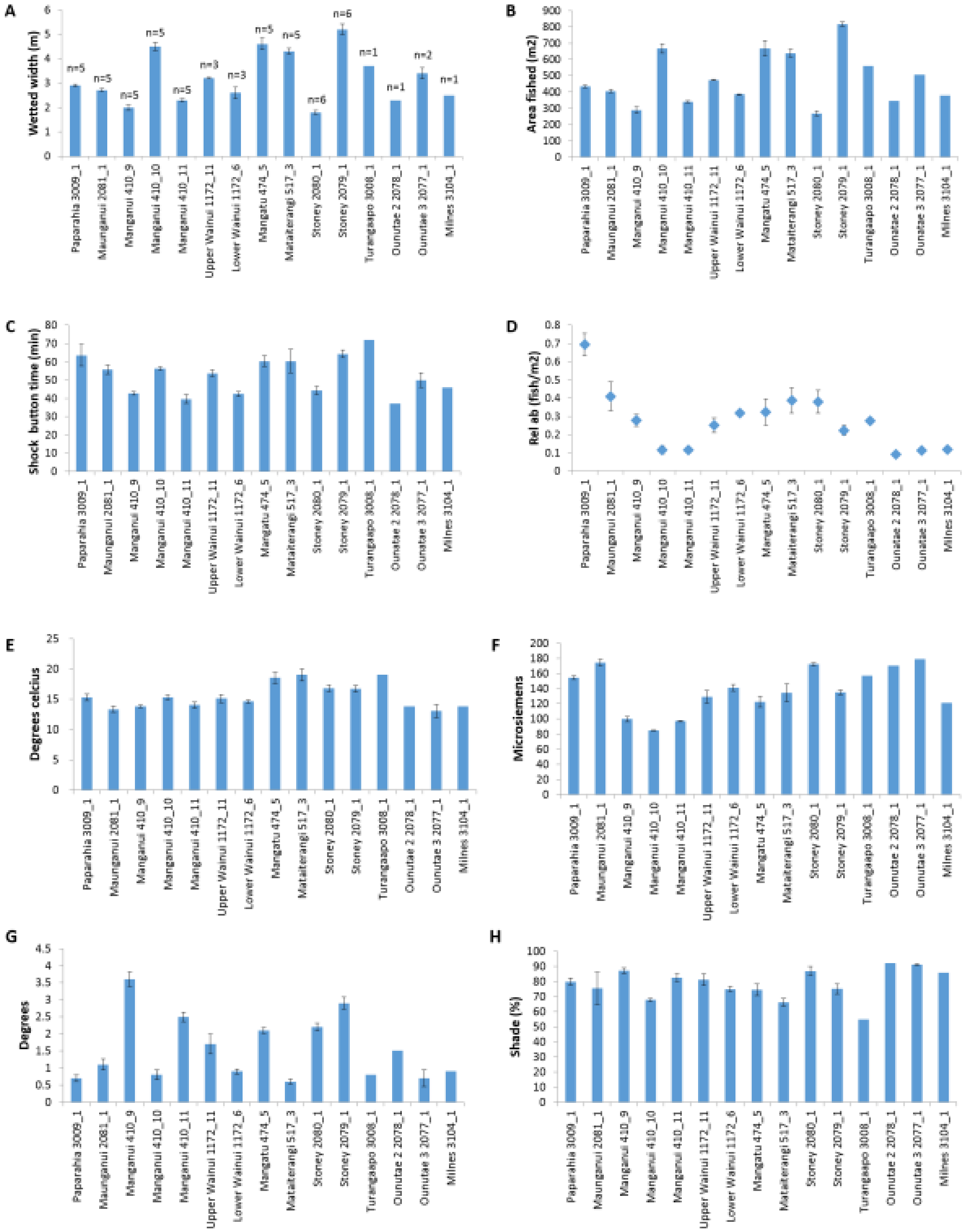


Figure 2: **Various parameters recorded at each of WRC's fish reference sites from 2009-2015. A) Average wetted width, B) Area electrofished, C) Electrofishing shock 'button' time, D) Relative abundance of fish, E) Stream temperature, F) Water conductivity G) Stream gradient, H) Stream shade. Variation expressed at +/- standard error. n = number of years each site has been sampled to generate data as displayed in A).**

Similarly the effort expended at each site when electrofishing was similar at each site over time (Fig. 2 C). During electrofishing the total amount of time that electrical charge is applied to the water is automatically aggregated by the machine when the button is depressed during fishing. This effort parameter is called 'button shock time'. At times the presence of more or less fish or slightly more or less water caused some slight variation in button shock time at a site. Button shock time across all reference sites ranged from an average of 35 - 70 min (Fig. 2 C)

The relative abundance of fish at each reference site was also surprisingly consistent over time when the entire community (including any newly recruited individuals) was considered (Fig. 2 D). Although some sites exhibited slightly higher variation in relative abundance than other sites, in general individual sites were relatively consistent over time. While it is acceptable to compare relative abundance of fish over time at the same site, comparing abundances between sites is problematic because various site specific parameters (e.g. conductivity, habitat complexity) can influence fishing efficiency. Relative abundances of fish recorded varied from approximately 10 fish/100m^2 (e.g. Manganui 410_10) to approximately 70 fish/100m^2 (e.g. Paprahia 3009_1). It is worth noting that the actual numbers of fish occupying the sampled reach at each reference site is likely to be somewhat higher than the values presented as they are based on single pass rather than depletion repeat pass electrofishing.

An alternative way of reporting fish abundance at a site is to express catches as the number of fish per 150 lineal meters of stream (Appendix. 6). Again for reference sites the number of fish per 150m of stream was relatively consistent within a site over time (Appendix. 6). Although area fished is ignored in this metric, it is worth noting that in many rivers fish may be disproportionately distributed within the wetted channel. That is, in many cases stream edges, where roots and undercuts may be more prevalent tend to support higher numbers and diversity than mid-channel areas which are often structurally less complex. Greater exploration of this metric for reporting will be considered in the future as more data are gathered.

Across all reference sites mean water temperatures ranged from an average of around 13^{0}C - 19^{0}C (Fig. 2 E) and for conductivity 85 -175 µs/cm (Fig. 2 F). Spot water temperature and conductivity measurements at the time of sampling were very similar at each reference site across years.

Stream gradient is measured for each 15m subreach using an inclinometer and two poles set at the top and bottom of a sub-reach (i.e. 15m apart). In general the average stream gradient (over 150m) was reasonably consistent at each site over time and mean reference site gradient ranged from between 0.7 - 3.7^0 (Fig. 2 G).

Shade was measured using a densiometer in the middle of the stream at every second subreach. A reading facing each of upstream, downstream, true left and true right was taken. The total number of filled squares (out of a total of 96) reflected by the canopy onto a convex mirror for each particular orientation was then averaged to give the shade at every second subreach. Shade for the entire site was represented as an average calculated from the average of the four readings at each subreach (4 readings x 5 subreaches = 20 values) Similar to most other parameters the average stream shade at individual reference sites was consistently similar over time with most sites maintaining >70% shade (Fig. 2 H).

Fish diversity at reference sites ranged from an average of just under 3 (Ounatae 3 2077_1) to just over 7 species (Paprahia 3009_1, Fig. 3).

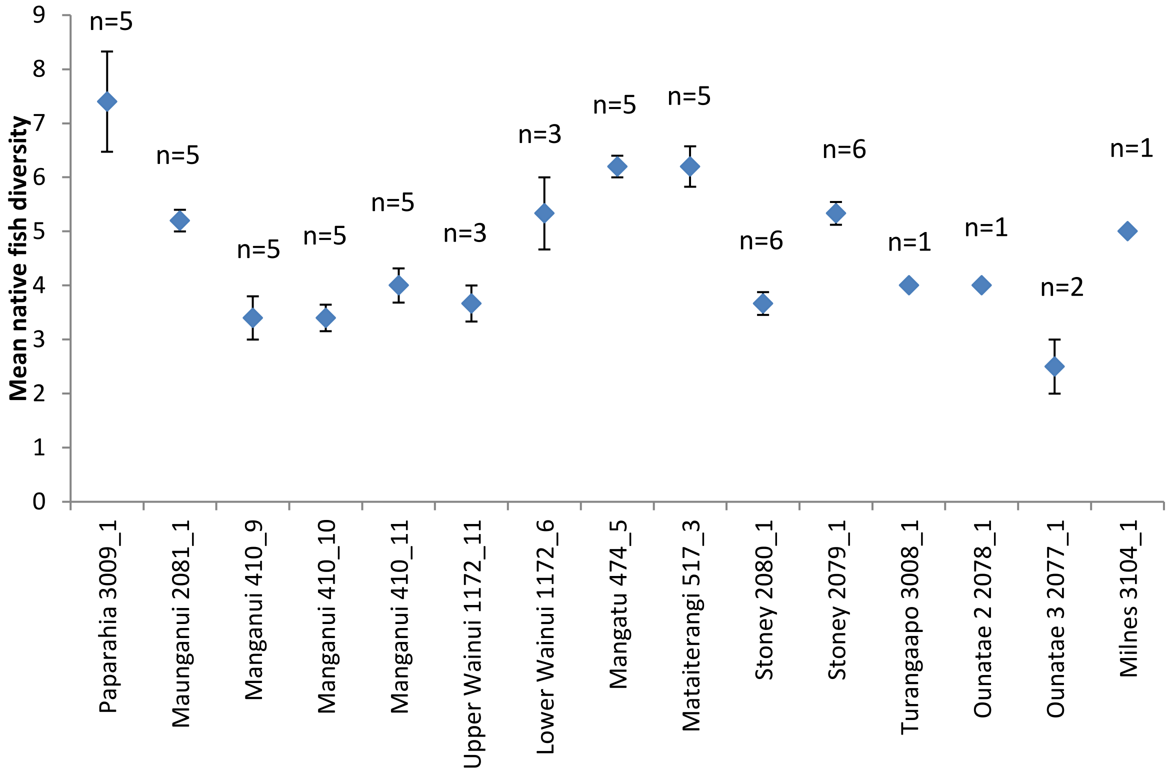


Figure 3: **Mean fish diversity at fish reference sites 2009 to 2015. Variation expressed at +/- standard error. n = number of years on which average is based for each site.**

Diversity was generally consistent at reference sites over time although at the most diverse site, Paprahia a gradual decline in diversity has occurred since sampling began (Fig. 3). The decline at this site is discussed further under 'Site specific assessments'.

An alternative way of using fish diversity for site assessment is to express it as part of an Index of Biological Integrity (IBI). A fish IBI was first pioneered in the United States (Karr 1981, 1987, Karr et al. 1986), and later modified and adopted for use in New Zealand (Joy and Death 2003, 2004). A Waikato region specific fish IBI was later developed (Joy 2006), and then refined further using a quantile regression approach (Joy 2007).

The quantile index takes into account and adjusts for fish diversity for distance inland and elevation, two variables known to be strong drivers explaining fish community composition in New Zealand. In addition to taxonomic richness, the index also incorporates three metrics related to habitat guilds used by different specialist fish species (riffle, benthic pool, pelagic pool), one metric related to tolerance to different environmental variables and one metric related to the proportion of native to exotic species present (excluding trout). As a guide to interpreting the final QIBI scores generated, the following score narratives have been provided in Joy 2007:

47 - 60 : defined as 'Excellent' and comparable to the best situations without human disturbance; all regionally expected species for the stream position are present. Site is above the 75th percentile of Waikato sites.

36 - 46 : defined as 'Good'. Site is above the 50th percentile of Waikato sites but species richness and habitat or migratory access reduced. Shows some signs of stress.

27 – 35 : ‘Moderate’. Site is above 25th percentile. Species richness is reduced. Habitat and or access is impaired.

6 - 26 : ‘Poor’. Site is impacted or migratory access almost non-existent.

0 : ‘No fish’ Site is grossly impacted or access non-existent.

All reference sites so far monitored have been classed on average as ‘Excellent’ (a score of 47 or greater) and reflect the minimally disturbed conditions at these sites (Fig. 4).

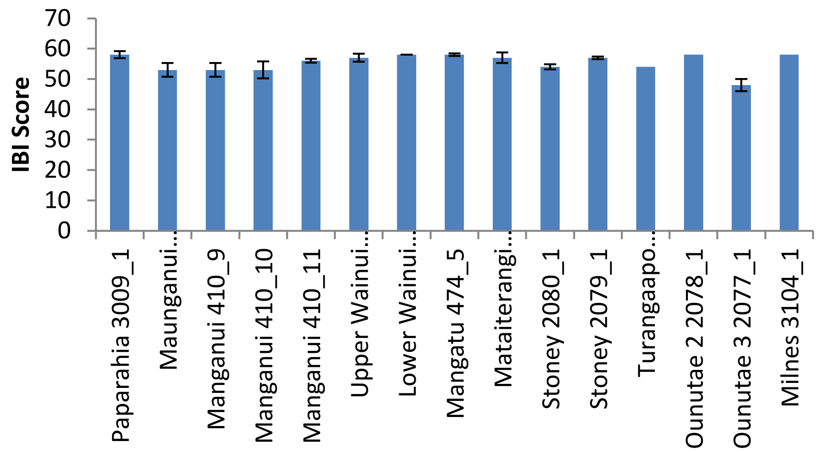


Figure 4: Average QIBI score for fish reference sites 2009-2015. Variation expressed at +/- standard error. See Fig. 3) for number of years on which average is based for each site.

3.1 Site specific assessments

This section profiles what appear to be some interesting and potentially emerging patterns or trends in the data collected from selected reference sites. The assessment is not exhaustive but provides a preliminary indication of the type of information that can be generated from the data to track directions in site specific and regional fish community attributes through time.

Paprahia stream

Paprahia stream on the west coast of the Waikato region is a high quality coastal reference site on private land. When first sampled in 2010, this stream had the highest native fish diversity of any reference site surveyed in the Waikato fish programme. Since this time there has been a decline in recorded diversity over time. Two species, lamprey (*Geotria australis*) and banded kokopu (*Galaxias fasciatus*) have declined both spatially and numerically to no detection. These two species were initially found within 4 different sub-reaches during the first sampling season (albeit in relatively low numbers) but over subsequent years, numbers declined and for the last 2 sampling years neither species has been detected at the site (Tables 2 & 3). A similar decline has been evident for torrentfish being present in two and four subreaches in 2010/2011 and 2011/2012 respectively. This species was not detected in 2012/2013 and although a single individual was captured in 2013/2014 none were subsequently detected in 2014/2015 (Table 4).

Table 2: **Spatial distribution of individual lamprey (*Geotria australis*) captured in the Paprahia stream over the last 5 years of monitoring. Letters A-J represent the same 10 continuous 15m subreaches (150m) sampled annually.**

GEO AUS	2010/11	2011/12	2012/13	2013/14	2014/15
A 15	1				
B 30	2	1			
C 45					
D 60	1	1			
E 75	1				
F 90					
G 105					
H 120		1	1		
I 135					
J 150					
Total	**5**	**3**	**1**	**0**	**0**
Subreach freq	**4**	**3**	**1**	**0**	**0**

Table 3: **Spatial distribution of individual banded kokopu (*Galaxias fasciatus*) captured in the Paprahia stream over the last 5 years of monitoring. Letters A-J represent the same 10 continuous 15m subreaches (150m) sampled annually.**

GAL FAS	2010/11	2011/12	2012/13	2013/14	2014/15
A 15	1	1			
B 30					
C 45			1		
D 60	1		1		
E 75					
F 90	2				
G 105			2		
H 120					
I 135					
J 150	2				
Total	**6**	**1**	**4**	**0**	**0**
Subreach freq	**4**	**1**	**3**	**0**	**0**

Table 4: **Spatial distribution of individual torrentfish (*Cheimarrichthys fosteri*) captured in the Paprahia stream over the last 5 years of monitoring. Letters A-J represent the same 10 continuous 15m subreaches (150m) sampled annually.**

CHE FOS	2010/11	2011/12	2012/13	2013/14	2014/15
A 15		1			
B 30	2				
C 45					
D 60					
E 75		1			
F 90					
G 105		1			
H 120					
I 135	1	2		1	
J 150					
Total	**3**	**5**	**0**	**1**	**0**
Subreach freq	**2**	**4**	**0**	**1**	**0**

In contrast, another species at this site, the bluegill bully (*Gobiomorphus hubbsi*), a species rarely encountered in the Waikato region, has consistently been detected in low numbers (1-3 individual fish) within the same one or two sub-reaches (E and or F) and the same riffles since sampling began (Table 5).

Table 5: Spatial distribution of individual bluegill bullies (*Gobiomorphus hubbsi*) captured in the Paprahia stream over the last 5 years of monitoring. Letters A-J represent the same 10 continuous 15m subreaches (150m) sampled annually.

GOB HUB	2010/11	2011/12	2012/13	2013/14	2014/15
A 15					
B 30					
C 45					
D 60					
E 75		2	1	2	2
F 90	1		2		1
G 105					
H 120					
I 135					
J 150					
Total	**1**	**2**	**3**	**2**	**3**
Subreach freq	**1**	**1**	**2**	**1**	**2**

During this same period, the most numerically and spatially dominant fish at this site has been the redfin bully (*Gobiomorphus huttoni*). This species has consistently dominated the assemblage with 200 or more individuals captured on each sampling occasion and 8 or more individuals being collected within each of the 10 subreaches since records began (Table 6).

Table 6: Spatial distribution of individual redfin bullies (*Gobiomorphus huttoni*) captured in the Paprahia stream over the last 5 years of monitoring. Letters A-J represent the same 10 continuous 15m subreaches (150m) sampled annually

GOB HUT	2010/11	2011/12	2012/13	2013/14	2014/15
A 15	28	26	32	24	25
B 30	40	15	16	22	17
C 45	31	24	22	30	33
D 60	36	13	39	24	26
E 75	36	20	31	32	31
F 90	11	11	15	12	22
G 105	31	26	35	29	18
H 120	27	19	17	30	8
I 135	22	23	17	19	14
J 150	32	27	23	22	18
Total	**294**	**204**	**247**	**244**	**212**
Subreach Freq	**10**	**10**	**10**	**10**	**10**

Manganui river and tributary reference sites

The Manganui River is a large tributary of the Awakino River. Three fish reference sites are located within the Manganui tributary network; a wider lower gradient 3rd order mainstem site in the upper reaches and two smaller forested higher gradient side streams (a 3rd and 2nd order) that enter the true left of the stream. Initial baseline sampling of the two Manganui side streams in 2010/11 documented native fish communities, made up of short and longfin eels, koaro, redfin bullies in the 2nd order stream, while in the 3rd order side stream shortjaw kokopu were also recorded. However in 2011/2012 a cohort of juvenile brown trout were detected in both stream reaches sampled with 28 (size range 76 – 111 mm) and 7 individuals (size range 77 - 109 mm) captured in the 2nd and 3rd order side streams respectively (Fig. 5A, B). The following year (2012/2013) trout numbers at both sites were lower but over subsequent years new recruitment has occurred. Over the same monitoring period the numbers of redfin bully have declined in both streams by more than 50%.

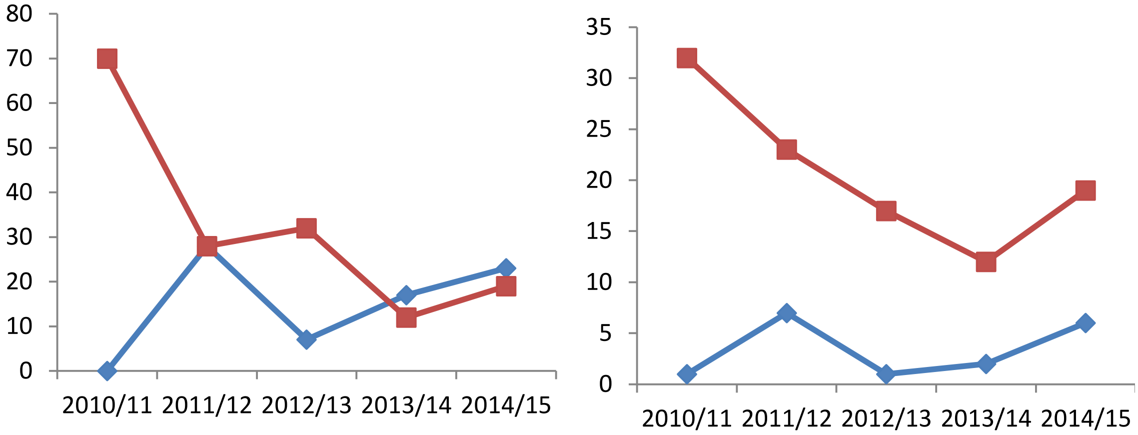


Figure 5A Numbers of redfin bully (red) and brown trout (blue) captured over time in the 2nd order (left) and 3rd order (right) Manganui tributary reference sites.

A closer look at redfin bullies

Although predation and or competition by trout on redfin bullies in the Manganui reference sites may have contributed to part of the redfin bully decline, on closer inspection an overall decline in abundance for redfin bullies has been evident throughout all west coast reference sites of the Waikato region since the programme began in 2010/2011 (Fig. 6). In several of these other sites no trout, or very low numbers, occur suggesting an alternative cause(s) for the numerical decline in this species. Interestingly the decline does not appear to be confined to west coast reference sites either. A small number of west coast random sites where redfin bullies were recorded, have similarly exhibited an apparent trajectory of decline between their three year rotation sampling (Fig. 7). In complete contrast, redfin bully numbers in every sampled reference and random site draining to the eastern coast of the Waikato region have increased over this same period (Fig. 6, 7).

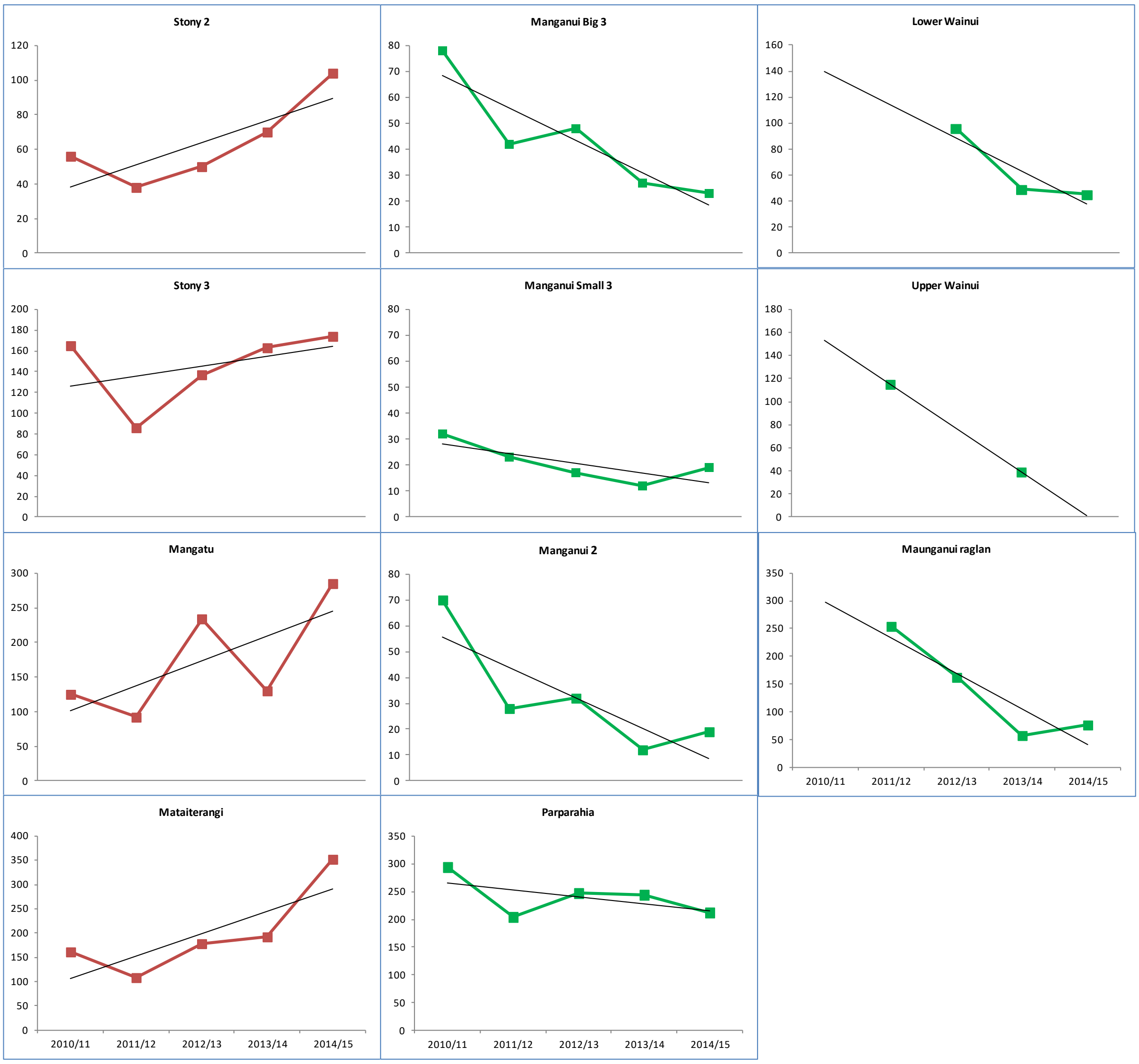


Figure 6: **Redfin bully numbers captured at each reference site from 2010/11-2014/15 in streams draining to the east coast (red panels n=4 sites) and west coast (green panels n=7 sites). Black line represents linear trend line. Note different Y axis scales for total fish numbers captured**

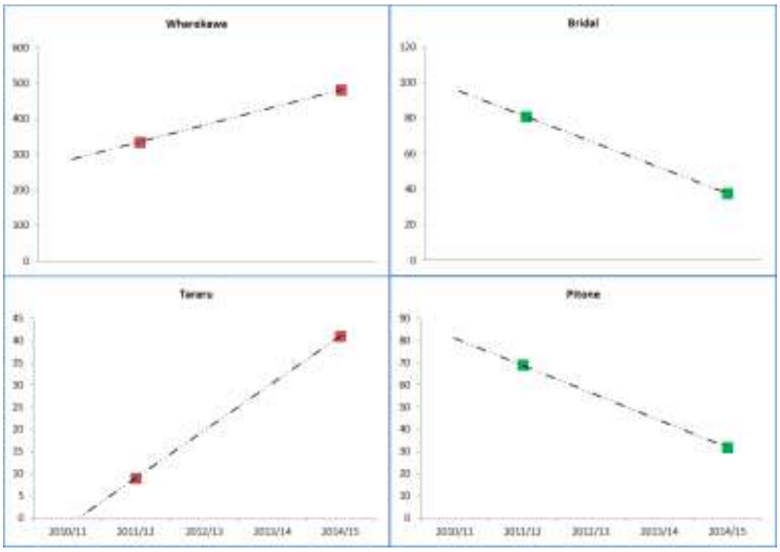


Figure 7: **Redfin bully numbers captured at each 3 year rotation random site in streams draining to the east coast (red panels n=2 sites) and west coast (green panels n=2 sites). Black dashed line represents early indicative trend direction. Note different Y axis scales for total fish numbers captured.**

Differences between redfin bully populations in streams draining to the east and west coasts do not appear to be confined to fish numbers either. When assessing the size structure of redfin bullies, it is apparent that a greater proportion of larger fish (particularly fish >70mm) are generally captured in streams (both reference and random) draining to the west coast compared to those draining to the east coast (Figs. 8, 9). Furthermore, the variability within a given size class also appears to be lower for this species in streams draining to the west. An annual (as opposed to mean) comparison of this same pattern within the same reference site network (over time) is presented in Appendix 3 and 4 for individual east and west coast sites respectively and for all east and west reference sites by year in Appendix 5.

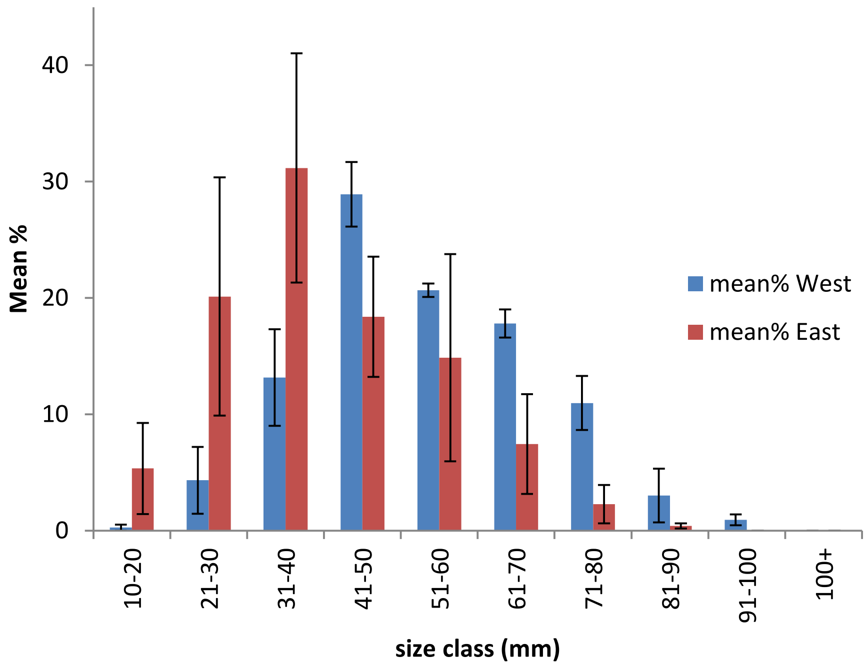


Figure 8: **Mean proportion of redfin bullies by size class from randomly fished sites where redfin bullies were found from 2011/12 - 2013/14 draining to the east coast (n=6 sites, 472 fish) and west coast (n=8 sites, 444 fish). Standard error within each size class (all sites, all fish) displayed.**

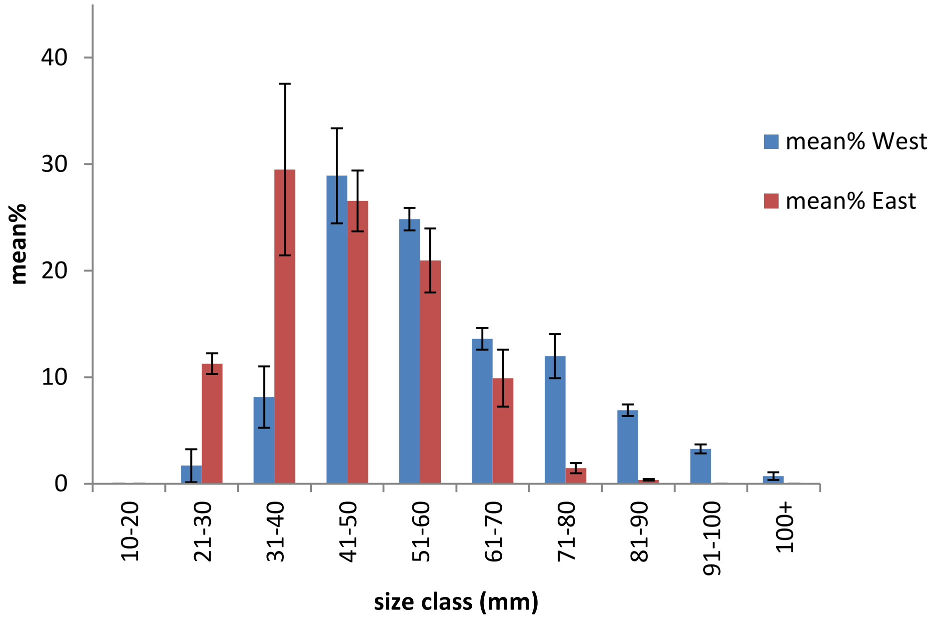


Figure 9: **Mean % of redfin bullies by size class from annually fished reference sites where redfin bullies were found from 2011/12-2013/14 draining to the East coast E (n= 4 sites, 1094 fish) and west coast (n=5 sites, 878 fish). Standard error within each size class (all sites, all fish) between rotation period displayed.**

In particular, redfins > 71mm have rarely been captured from any east coast random or reference site, whereas in sampled west coast streams more than 20% of the redfin population may be represented by fish >71mm. In virtually all cases, recruitment of redfin bullies in the west coast draining sites appears to have been very weak over the last 5 years while recruitment in the east has been strong.

3.2 Random sites 2011/2012 v 2014/2015

The first return visit to 39 random sites first sampled in 2011/2012 occurred in summer 2014/15. Average values for a range of parameters were typically similar between this time period but notable differences included a general increase in mean conductivity and a general decrease in the average relative abundance of fish at random sites (Table 7). Other notable differences included an increase in the total number of sites where no fish were present (from 7 to 10) and an increase in the total number of sites with no fish and no koura (from 2 to 6) (Table 7).

Table 7: **Summary of regional parameters calculated from the same 39 random sites fished in 2011/2012 and 2014/2015. Values in brackets represent range.**

Random sites (n=39 sites)	2011/2012	2014/2015
Total button shock time (min)	1401	1400
Average shock time (min)	35.92 (11-81)	35.89 (15-91)
Average water temperature (C)	16.48 (11.8-22.9)	16.17 (11.0-20.6)
Average conductivity (µs/cm)	123.8 (46-294)	143.7 (50-321)
Average wetted width (m)	2.35 (0.4-8.5)	2.38 (0.6-7.8)
Average stream gradient (deg)	1.21 (0.1-5.2)	1.18 (0.2-3.8)
Average stream shade (%)	32.3 (0.0-99.5)	29.9 (0.0-93.0)
Average species diversity	2.48 (0-10)	2.44 (0-9)

Average native species diversity	2.05 (0-8)	2.03 (0-8)
Average introduced species diversity	0.44 (0-5)	0.41 (0-3)
Average fish IBI score (actual/60)	28.36 (0-60)	27.9 (0-60)
Average relative ab all (fish/100m2)	24.42 (0-314.6)	17.09 (0-142.8)
Average relative ab native (fish/100m2)	23.14 (0-314.6)	15.64 (0-142.8)
Average relative ab introduced (fish/100m2)	1.28 (0-27.2)	1.45 (0-23.1)
Number of sites with koura but no fish	6	4
Total sites with no fish	7	10
Total sites with no fish and no koura	2	6
Total fish diversity	18	20
Total native diversity	11	13
Total exotic diversity	7	7

In both 2011/12 and 2014/15 shortfin and longfin eels were the most widely distributed fish species being present at 69% (27/39) and 56% (22/39) of the random sites respectively in 2011/12 (Fig. 10) and at 67% (26/39) and 51% (20/39) of the sites in 2014/15 (Fig. 11). A total of 18 and 20 different fish species were recorded during these two sampling periods comprising 11 native and 7 introduced species in 2011/12 and 13 native and 7 introduced species in 2014/15.

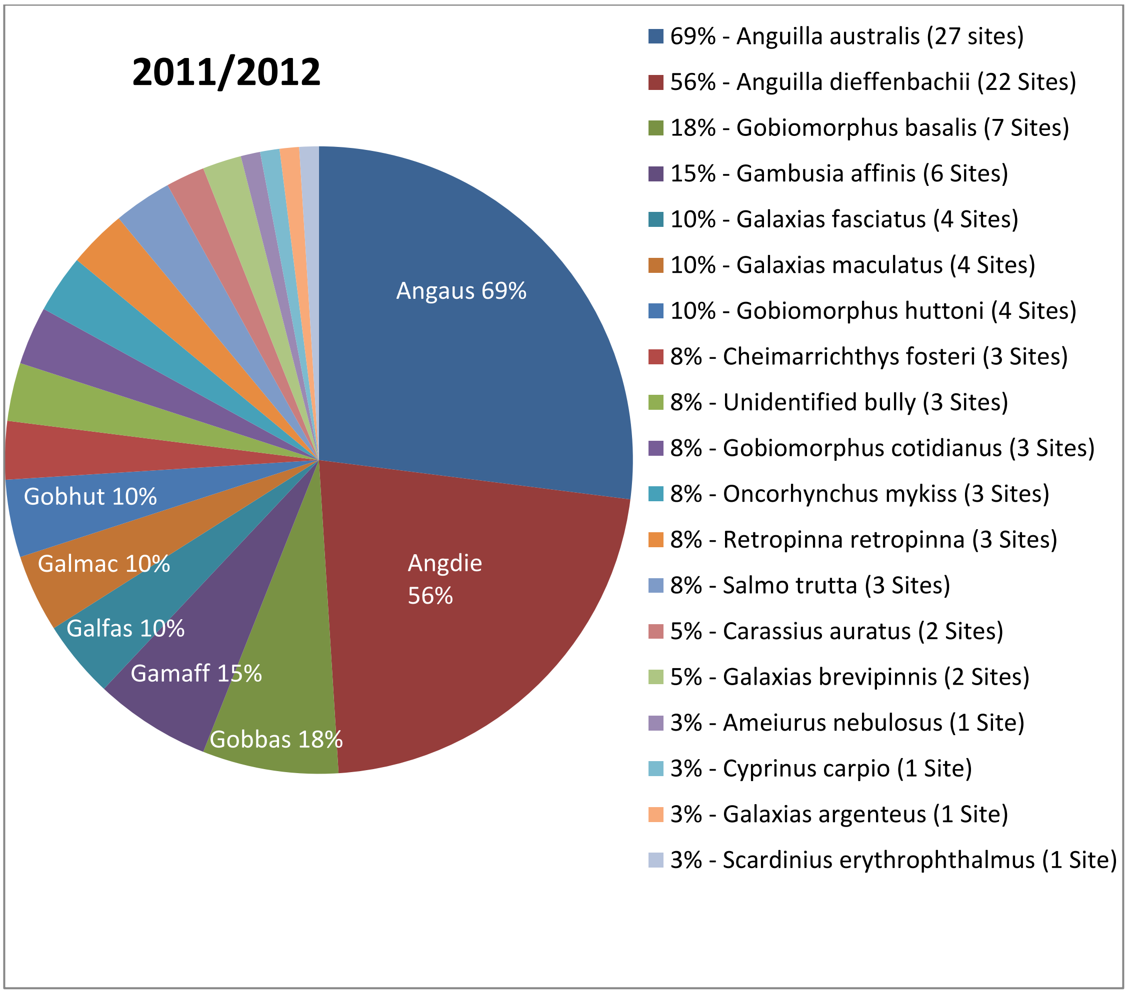


Figure 10: Pie chart depicting regional commonality of individual fish species detected at the 39 random sites sampled in 2011/2012. Those individual species detected at 10% or more of sites indicated as white text on wedges

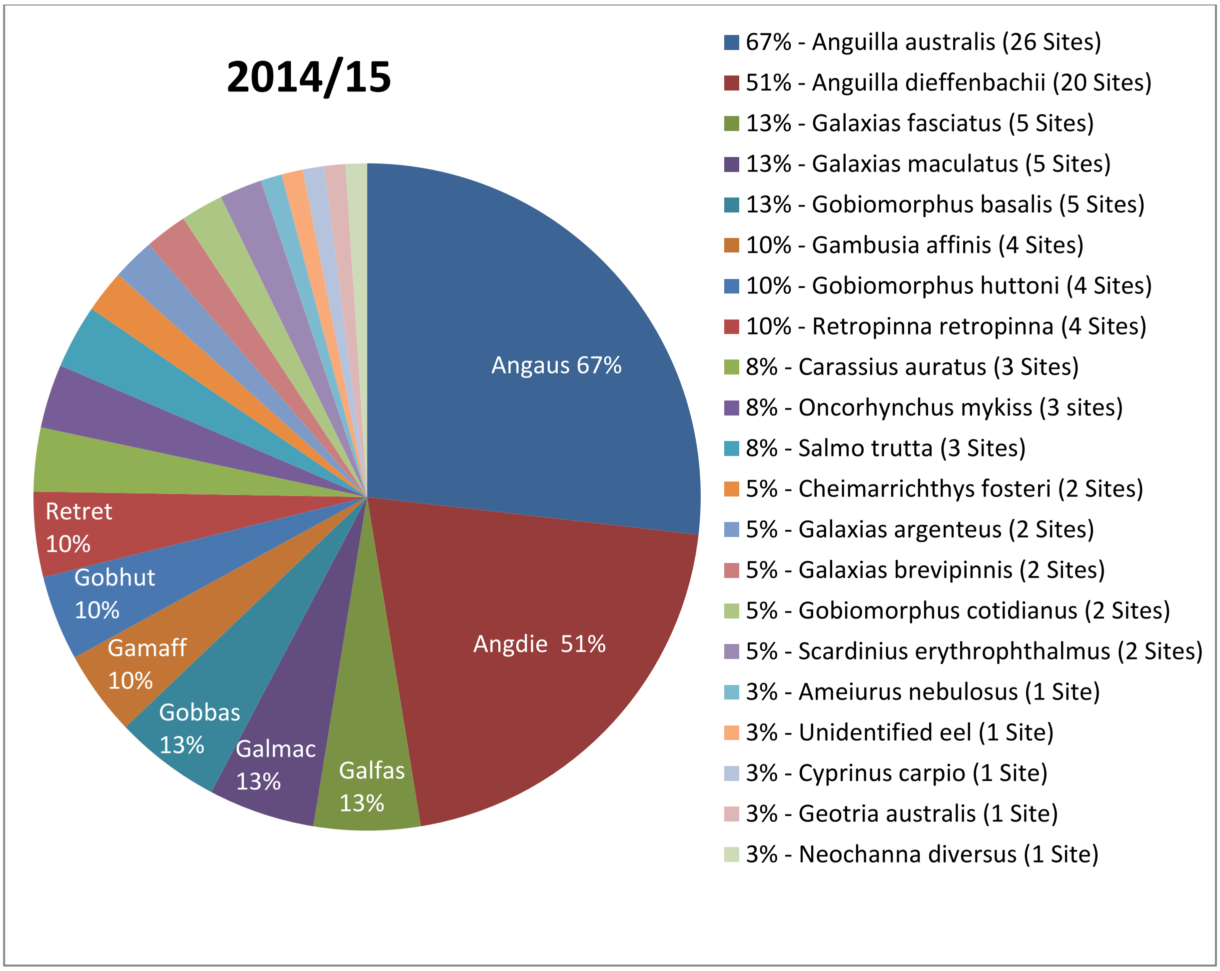


Figure 11: Pie chart depicting regional commonality of individual fish species detected at the 39 random sites sampled in 2014/2015. Species which were recorded from at least 10% of sites are indicated as white text on wedges

Across the 39 monitored sites there was little change among species presence/absence with at most a +/- 2 site increase or decrease in detection for some species and no change for others (Table 8). Similarly there was little change to the regional distributional rank order among species during this time period (Table 8).

Table 8: Proportion of random sites (electrofishing only) where different fish species were detected in 2011/2012 versus those same sites in 2014/2015 (numbers in brackets represent the number of sites where the species was detected). Distribution rank represents most common to least commonly encountered. Species shaded grey are introduced. Green = increase, orange = decrease, blue = no change in detection.

Fish	%Random Sites detected		3 yr site change	Distribution rank	
Species	2011/2012 (39)	2014/2015 (39)	+/-	2011/2012	2014/2015
Shortfin eel	69 (27)	67 (26)	-1	1	1
Longfin eel	56 (22)	51 (20)	-2	2	2
Crans bully	18 (7)	13 (5)	-2	3	3
Gambusia	15 (6)	10 (4)	-2	4	6
Banded kokopu	10 (4)	13 (5)	1	5	3
Inanga	10 (4)	13 (5)	1	5	3
Redfin bully	10 (4)	10 (4)	0	5	6
Torrentfish	8 (3)	5 (2)	-1	8	12
Common bully	8 (3)	5 (2)	-1	8	12
Rainbow trout	8 (3)	8 (3)	0	8	8
Brown trout	8 (3)	8 (3)	0	8	8
Common smelt	8 (3)	8 (3)	0	8	8
Goldfish	5 (2)	8 (3)	1	13	8
Koaro	5 (2)	5 (2)	0	13	12
Catfish	3 (1)	3 (1)	0	15	17
Common carp	3 (1)	3 (1)	0	15	17
Giant kokopu	3 (1)	5 (2)	1	15	12
Rudd	3 (1)	5 (2)	1	15	12
Black mudfish	0 (0)	3 (1)	1	19	17
Lamprey	0 (0)	3 (1)	1	19	17

3.3 Extent estimates

The probabilistic network design in unison with the R package SPSURVEY, (see Collier & Hamer 2012 for detailed explanation), can be used to estimate the extent to which categories and values of ecosystem metrics occur across a target river network. In this preliminary example we have used the fish QIBI metric to demonstrate how the network can be used to extrapolate the IBI across the entire regional wadeable river network on developed land. For instance based on data gathered for fish from 2012-2014, it is estimated that around 21% of the Waikato Region's wadeable river network on developed land would support fish communities with an 'excellent' IBI score, equating to around 2100km of the network (Table 9.). In contrast over half of the network is predicted to exhibit 'moderate - poor' IBI scores with a further 14 % of the network likely to have no fish (Table 9). These estimates are based on both electrofishing and netting data collected over 3 years (2012-2014) at 144 individual random river sites across the region. Most of these data are driven by electrofishing data (123 sites) but a complete breakdown of these extent estimates by method type (electrofishing and netting) can be found in Appendix 7.

Table 9: **Fish QIBI extent estimates for the Waikato region wadeable river network**

QIBI Category	Est % network	StdError	Est network km	StdError
Excellent	21.15	3.09	2109.82	318.80
Good	18.80	2.88	1874.86	279.87
Moderate	28.74	3.62	2866.75	377.26
Poor	16.97	3.27	1693.18	348.14
No Fish	14.34	3.06	1430.24	326.93

The regional average estimated fish IBI score (both methods) was calculated to be 31.4 (moderate – Table 10). With trapped sites having a slightly higher average QIBI score.

Table 10: Average fish QIBI score estimates for the Waikato region wadeable river network for both methods combined and individually. Standard error and upper and lower 95% confidence boundaries indicated.

Method	QIBI mean est	StdError	LCB95Pct	UCB95Pct
Both	31.39	1.45	28.56	34.24
EFM	30.95	1.57	27.87	34.04
Trapping	35.23	3.45	28.47	41.99

3.4 Reference sites 2011/2012 v 2014/2015

Nine reference sites were fished in 2011/2012 and then again in 2014/2015. Most parameters were very similar between these sampling periods. Notable differences included an on average overall higher relative abundance of fish recorded in 2014/2015 (38.4 fish/100m^2) compared to 2011/2012 (26.4 fish/100m^2). This increase appears to be due to a proportionally higher increase in capture of native fish since the average capture of invasive fish declined slightly during the same period (Table 11).

Table 11: Summary of regional parameters calculated from the same 9 reference sites fished in 2011/2012 and 2014/2015. Values in brackets represent range.

Reference sites (n= 9 sites)	2011/2012	2014/2015
Total button shock time (min)	429	510
Average shock time (min)	47.7 (36-69)	56.7 (43-67)
Average water temperature (C)	15.3 (13.6-16.7)	16.5 (13.9-21)
Average conductivity (µs/cm)	126 (85.4-170.4)	134 (85.6-188.4)
Average wetted width (m)	3.47 (2.0-5.6)	3.28 (1.8-4.8)
Average stream gradient (deg)	1.81 (0.4-3.8)	1.98 (0.8-3.7)
Average stream shade (%)	76.3 (59.8-92.1)	76.2 (69.5-83.7)
Average species diversity	4.88 (2-9)	4.88 (3-7)
Average native species diversity	4.66 (2-9)	4.88 (3-7)
Average introduced species diversity	0.55 (0-2)	0.44 (0-2)
Average fish IBI score (actual/60)	55.6 (50-60)	56.2 (52-58)

Average relative ab all (fish/100m2)	26.4 (13.4-65.2)	38.4 (8.3-61.6)
Average relative ab native (fish/100m2)	24.9 (10.6-64.6)	36.8 (10.4-61.7)
Average relative ab introduced (fish/100m2)	1.45 (0-8.7)	0.94 (0-6.7)
Number of sites with koura but no fish	0	0
Total sites with no fish	0	0
Total sites with no fish and no koura	0	0

3.5 Random vs Reference fish QIBI 2010 – 2015 and 2012 v 2015

The average fish QIBI score for the 39 random sites fished in 2012 was very similar to that recorded in 2015 from the same sites with median values being similar (QIBI = 34 in 2012 and 36 in 2015, Fig. 12). The 9 reference sites also exhibited similar QIBI means and identical median scores between sampling periods. QIBI scores were significantly higher at reference sites (QIBI mean =55.6 in 2012 and 56.2 in 2015, median=56 both periods) and the variance much lower than those calculated for random sites. A similar difference in QIBI scores was calculated between reference and random sites irrespective of year of sampling (Fig. 12)

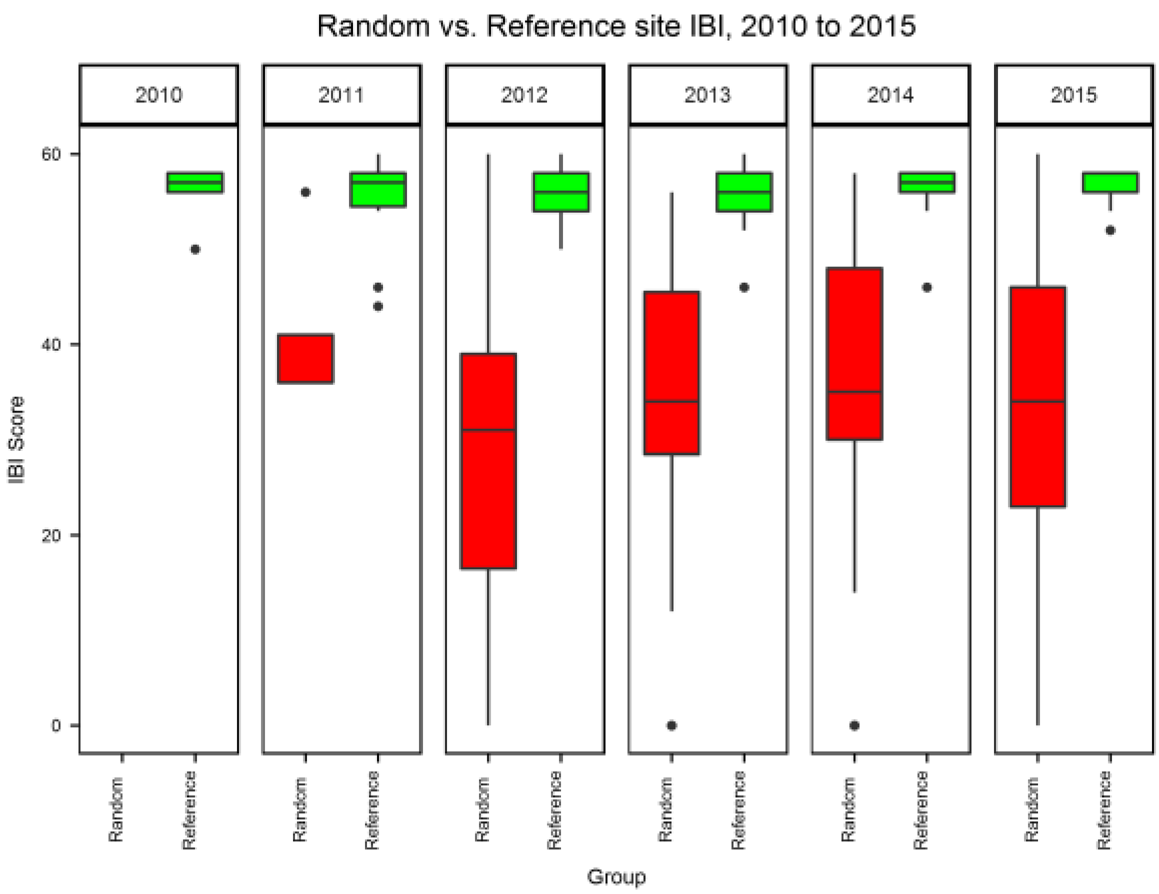


Figure 12: **Fish IBI scores for each year of assessment comparing scores derived from random and reference (minimally impaired) sites since inception of programme (electrofishing sites only). Note: Only reference sites were sampled prior to 2011, hence the absence of data for random sites in 2010. Upper and lower hinges represent 25th and 75th percentiles respectively and median line represented.**

A closer look at eels

Short and longfin eels were regularly captured at more than half of the sites sampled. An interesting regional pattern that seems to be emerging from data collected during the course of this programme is that shortfin eels tend to be more commonly encountered inland (>75km) compared to longfins (Fig. 13 top and bottom graphics respectively). Also, small shortfin elvers (<100mm) appear to be more commonly encountered at distances greater than 75km from the coast compared to longfin elvers, which tended to be more regularly encountered closer to the coast (Fig. 13 top and bottom graphics). Overall, longfin elvers (<100mm) only made up a very small proportion of all longfins captured.

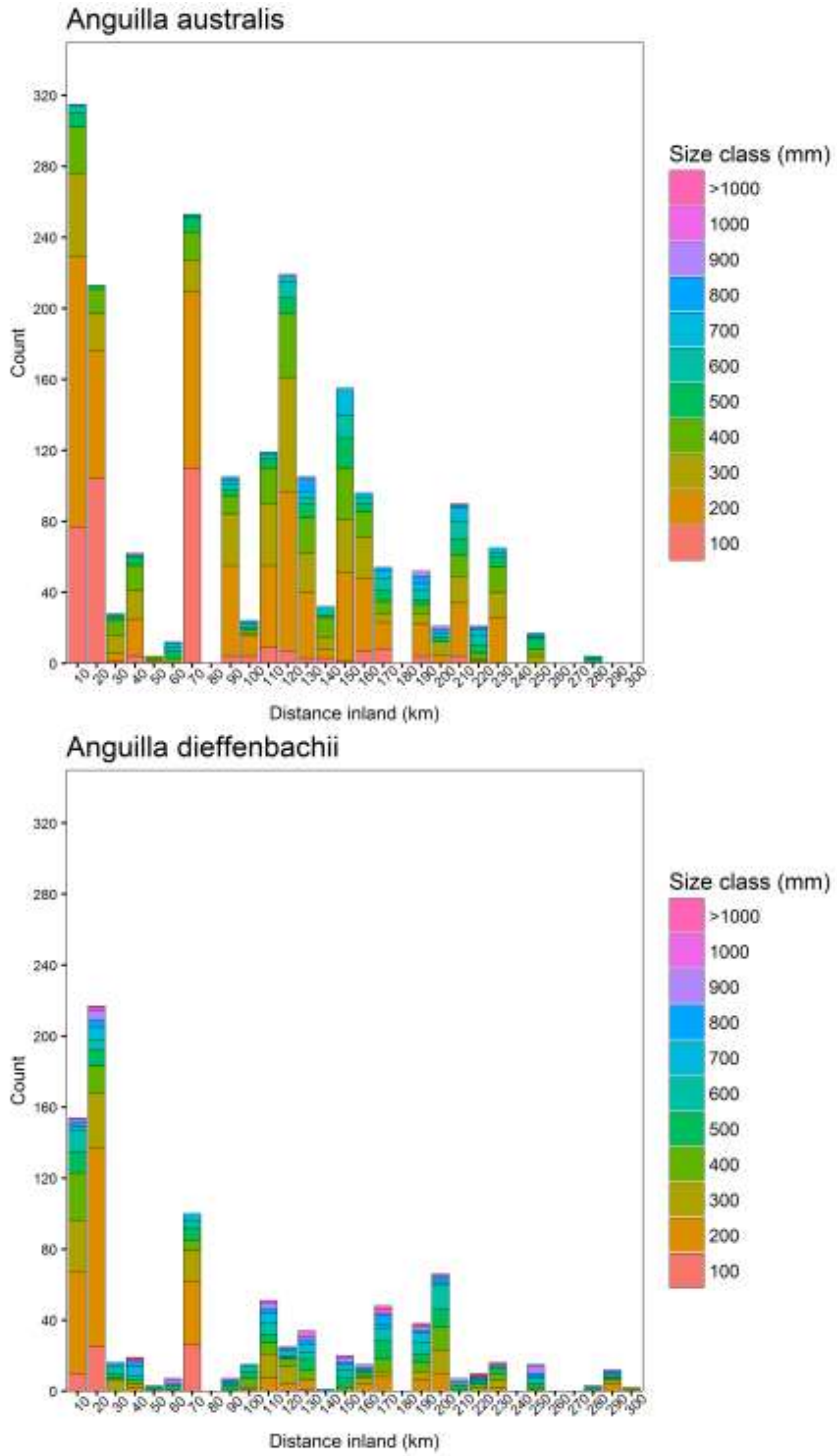


Figure 13 **Distribution of shortfin (above) and longfin (below) eel lengths by distance inland for all random sites and all years (electrofishing data only)**

This pattern appears to exist when data for all sites in all years are plotted. Interestingly the same difference between longfins and shortfins for distance inland was evident in the same 39 random sites sampled 3 years apart (Figs. 14). That is, the likelihood of encountering shortfins rather than longfins further inland does not at this stage appear to be influenced by year. Further rotations of data will help to establish the strength of this observation.

Another interesting observation is that at many sites where the existing adult eel population may be dominated exclusively or almost exclusively by longfins, the majority of small eels (< 100mm) tended to be shortfins. An example of this over time can be seen in Appendix 8 for reference sites.

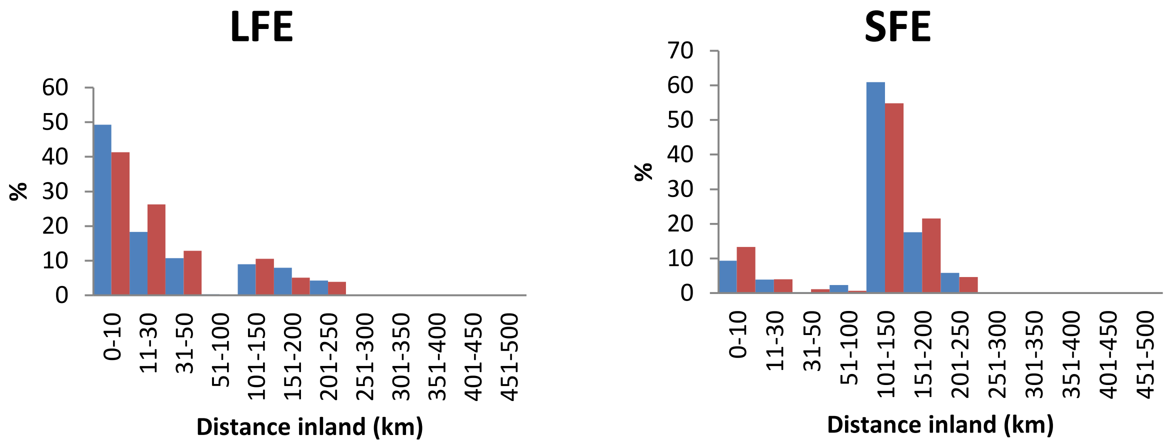


Figure 14: **Proportion of longfin eels (left) and shortfin (right) captured for distance inland from all reference (n=9) and random sites (n=39) fished in 2012 (blue bars) and then for the same sites fished 3 years later in 2015 (red bars).**

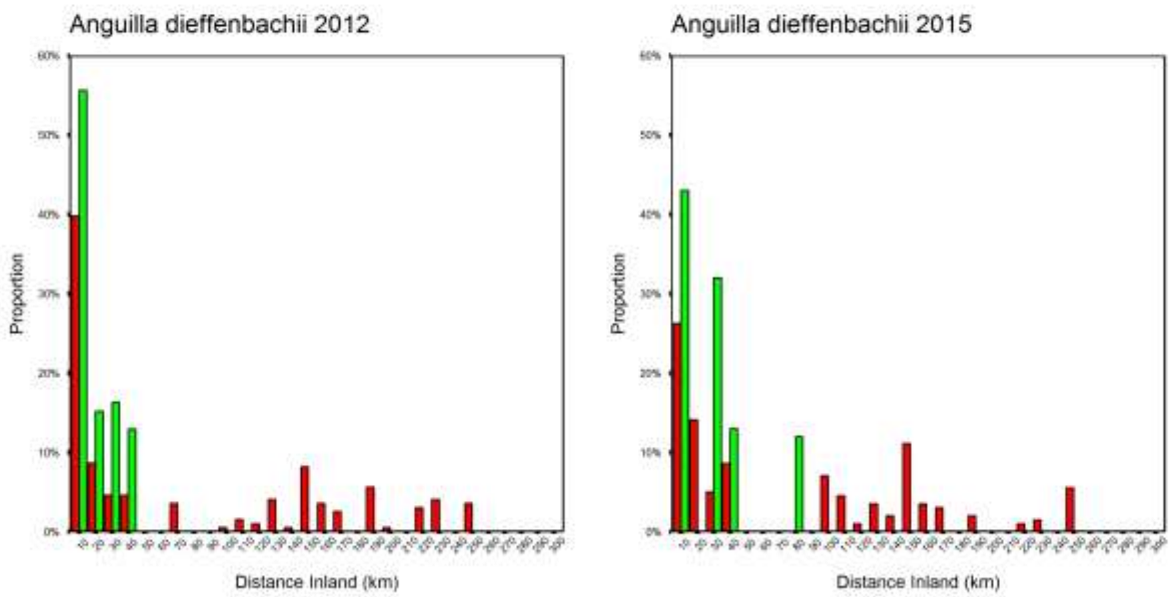


Figure 15: **Proportion of longfin eels by distance inland captured from reference (green bars) and random (red bars) sites in 2012 (left) and the same set of sites in 2015 (right). Electrofishing sites only (n= 9 reference, n= 39 random)**

As noted earlier an acknowledged shortcoming of the reference site network is that unimpaired inland sites (>80km) are difficult to find. Nevertheless, when reference and random sites sampled in 2012 were sampled again in 2015, it is apparent that even at random sites a large proportion (c. 50%) of all longfins captured across both time periods were close to the coast (< 50km, Fig. 15).

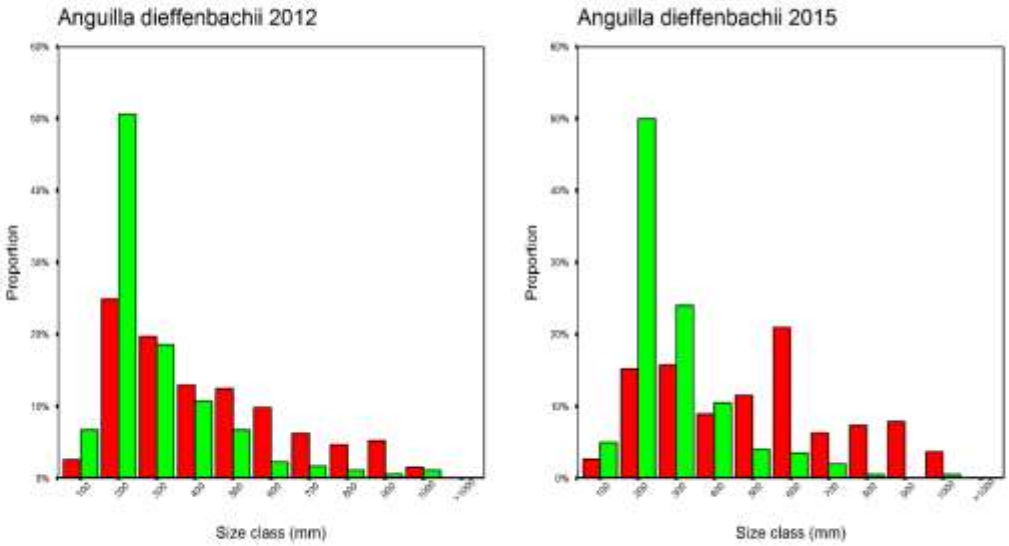


Figure 16: **Proportion of longfin eels by size class captured from reference (green bars) and random (red bars) sites in 2012 (left) and the same set of sites in 2015 (right). Electrofishing data only.**

The proportion of longfin eels by size class at minimally impaired references sites is generally consistent over time and such streams do not appear to support many longfins of 'harvestable' size despite the absence of harvest pressure (Fig. 16). Longfin eels between 100-300mm represented the greatest proportion of captures. In contrast the size structure at random sites appeared to be more variable between time periods with a greater proportion of harvestable longfins present, particularly in 2015 (Fig. 16). Very few longfin elvers (<100mm) were captured in 2012 or 2015 irrespective of whether sites were random or reference. A low proportion of longfin elvers (<100mm) in the overall longfin catch from random sites has been noted since the beginning of the programme (Fig. 13 bottom graphic). In contrast, while longfin eels are the dominant anguillid at most reference sites, it is interesting to note that small shortfin eels (up to 200mm) comprised the vast majority of the shortfin catch at those sites, while at random sites a greater size range was captured (Fig. 17)

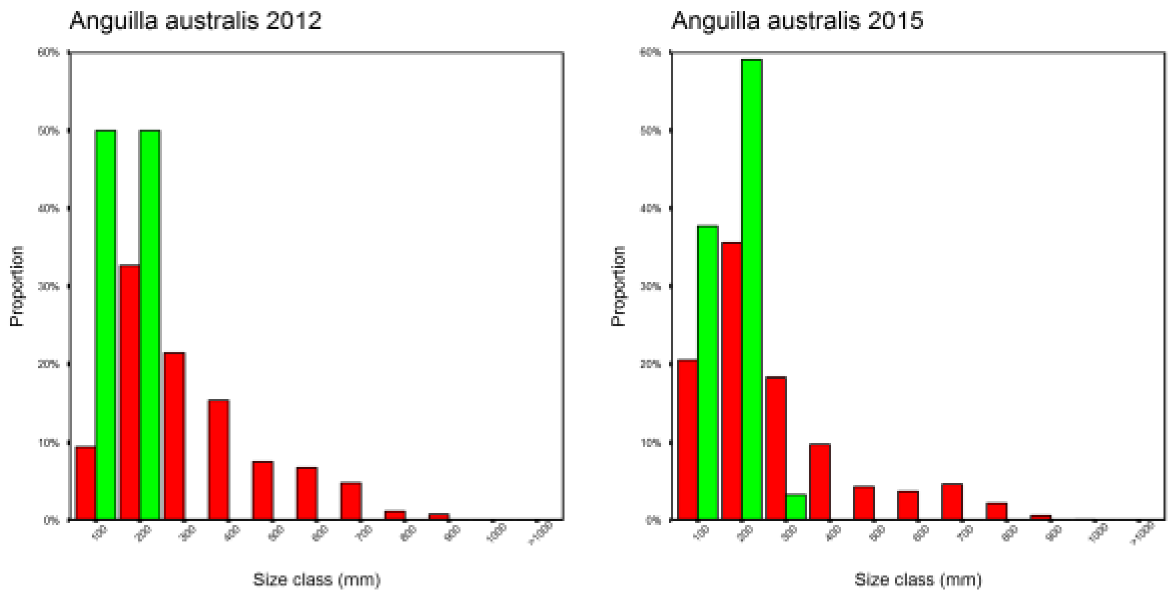


Figure 17: **Proportion of shortfin eels by size class captured from reference (green bars) and random (red bars) sites in 2012 (left) and the same set of sites in 2015 (right). Electrofishing data only.**

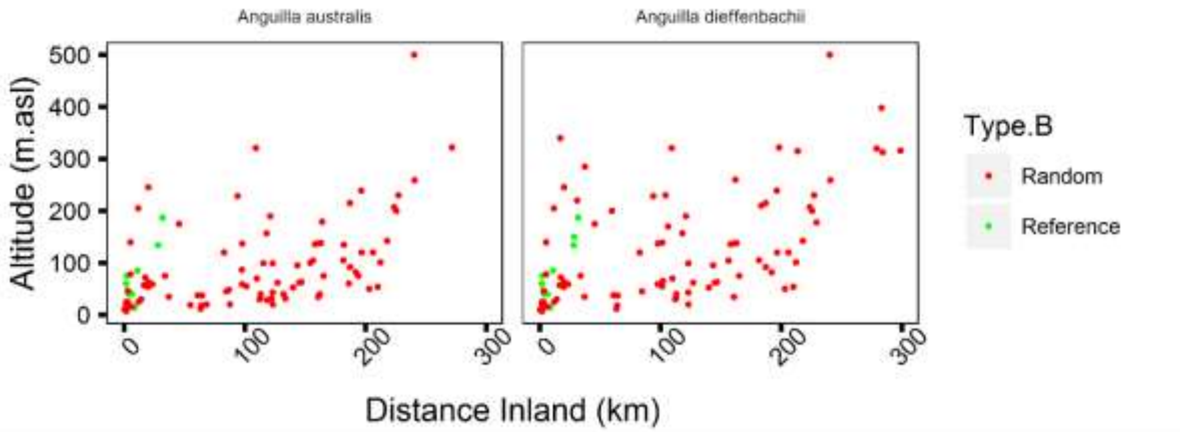
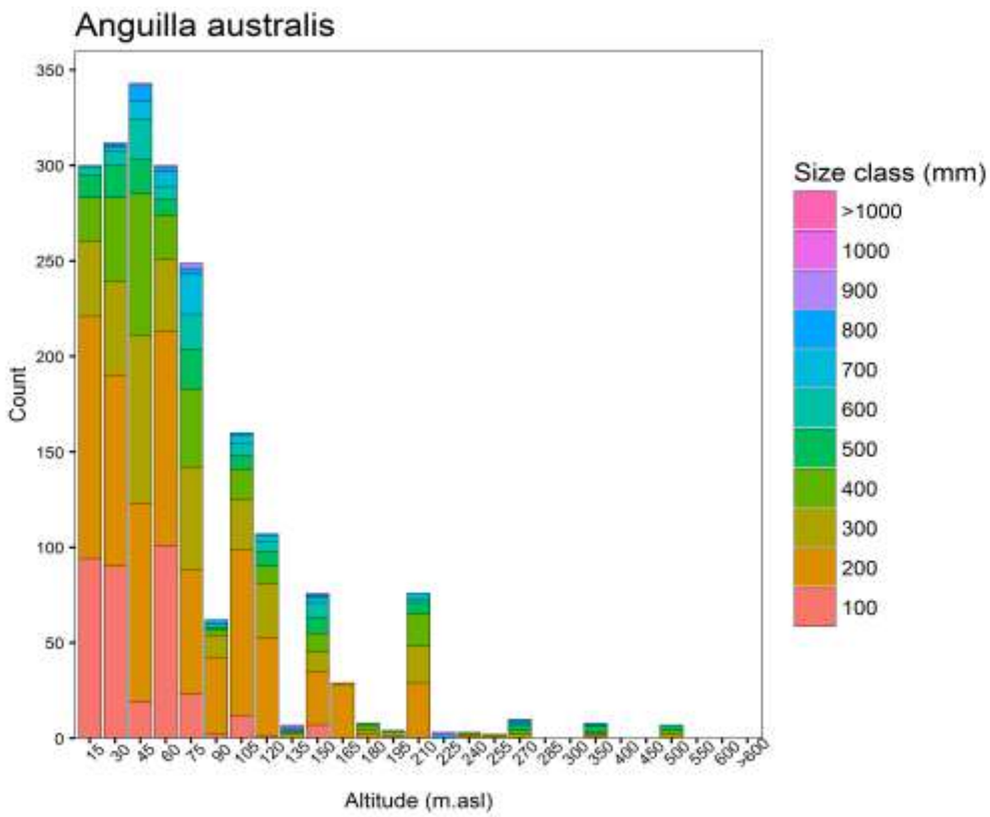


Figure 18: **Distribution of shortfin (left) and longfin (right) eels by altitude and distance inland from the Random (red) and Reference (green) network 2012-2014.**

In terms of presence or absence, there does not appear to be much difference in the distribution of short and longfin eels by altitude and distance inland within the Waikato region (Fig. 18). There were a few more sites at high altitude and further inland where only longfin eels were present but numbers of longfin at these sites were typically very low as illustrated in Fig. 19 (altitude, bottom panel) and Fig 13. (distance inland, bottom panel) respectively.



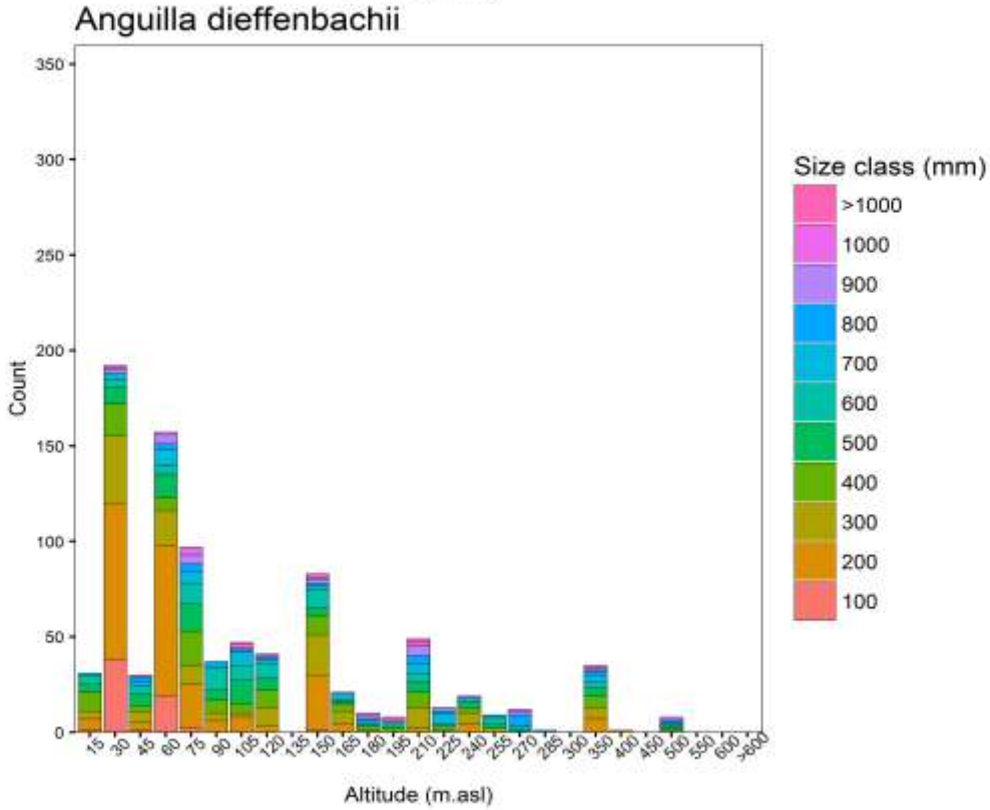


Figure 19: Distribution of shortfin (above) and longfin (below) eel lengths by altitude for all random sites and all years (electrofishing data only) within the random network.

With respect to sampling protocol there does appear to be an apparent difference in eel size recorded whereby the mean and median size of eels captured via electrofishing is noticeably lower than that by netting over the same spatial scale (Fig. 20). To some degree this apparent difference may reflect the generally deeper (but wadeable) habitats that are more conducive to netting.

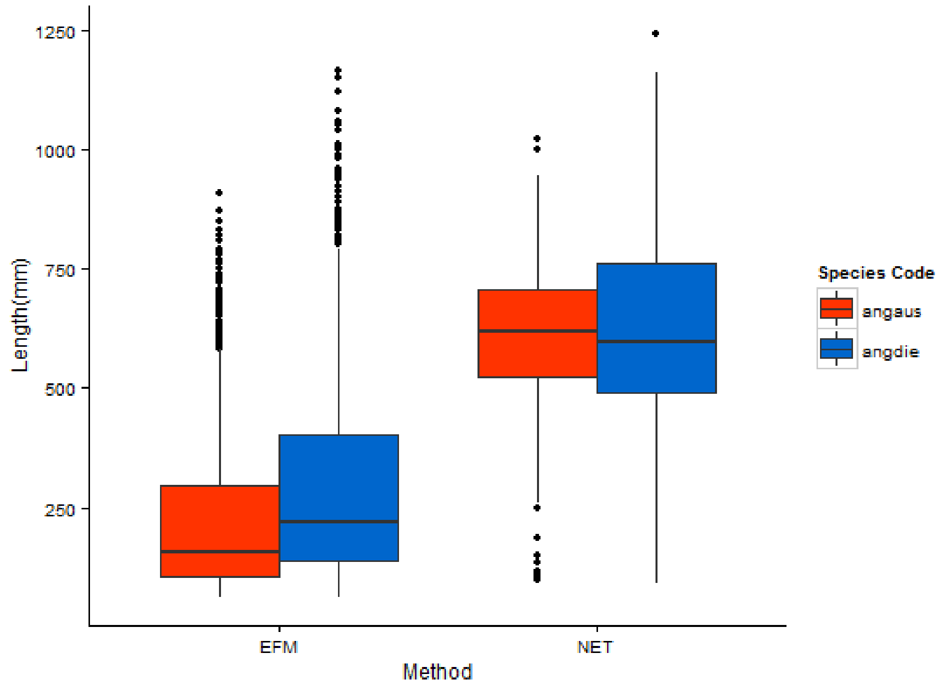


Figure 20: **Boxplots of shortfin (red) and longfin (blue) eels captured by either electrofishing (n=124 sites) or netting (n= 21 sites) protocols throughout the programme.**

4 Discussion

Ecological community structure and function in freshwater environments is complex and inherently prone to change both spatially and temporally. In the Waikato region, a wide range of river ecosystems exist which can vary greatly with respect to climate, underlying geology and influence of human pressure (Collier et al. 2010). To complicate matters further, native fish populations in the Waikato typically comprise numerous species with diadromous life-histories. Consequently while fish assemblages are being assessed at a reach scale, it is quite possible that population characteristics such as recruitment strength in any given year may be influenced by factors or conditions occurring at much larger spatial scales. When applying methodologies to assess fish communities it is paramount to minimise methodological variation or 'noise' as much as possible to increase the likelihood of detecting any trends that may be occurring within the system of interest. It is also likely that information collected using the same methods in other regions would greatly assist interpretation of regional reach scale data for native fish communities. For example, it is possible that recruitment of some diadromous fish species to river systems within the Waikato region may be originating from larval outputs generated from catchments in other regions.

To improve our understanding of local, regional and national scale processes (natural and/or human mediated) influencing fish assemblages across the diverse array of wadeable stream types, consideration of both methodology and network design is critical. This report provides the first assessment of the state of fish communities in wadeable streams within the Waikato region. The use of standardised sampling protocols in association with a probabilistic network design (and an electronic field data capture system) will enable an unbiased and robust preliminary baseline of reach scale fish community attributes against which future sampling and results can be rigorously and defensibly evaluated.

In undertaking this first assessment, the general approach was to first assess the longest running and least impaired annually monitored reference sites, and then to compare data from these sites with those collected at random sites following the first 3 year sampling rotation of the network. It is acknowledged that insufficient wadeable reference sites for fish are currently monitored (and available) to cover the variety of wadeable river types in the region. In particular there are limited minimally impaired sites at low elevation and >30km inland. At present this problem constrains our ability to effectively assess the present state of low elevation inland fish communities under varying levels of anthropogenic influence. To address this shortcoming, future work to investigate the potential to reconstruct likely pre European/Maori fish communities through modeling and use of historical information and data sources (e.g. historic vegetation layers) is proposed.

4.1 Protocol consistency assessment

Reference sites

Notwithstanding issues of representativeness noted above, reference site assessment included an evaluation of variability in recorded methodological parameters between years as well as various fish community attributes derived from data collection. Encouragingly there was minimal variation between years for various repeated methodological parameters measured at individual sites within the reference network. In particular the variation in parameters such as electrofishing machine button 'shock' time, wetted area, area fished, gradient, shade and conductivity within each site over time was low. Low variation in these methodological parameters reduces 'data noise' and improves the potential for evaluating any patterns or trends that may have occurred to the fish community since previous assessment(s).

With respect to the fish communities at reference sites, a range of community attributes such as the relative abundance, diversity and proportional balance (of the assemblage) of fish species were generally consistent and relatively stable at individual sites over time. There were some notable changes within selected reference sites over this time period however. For instance, the gradual decline in fish species richness at the west coast Paparahia stream is notable. At this relatively early stage (5 years data) it is difficult to ascertain the likely mechanisms behind the decline in richness at this site but it is possible that mechanisms underlying the change may differ and may be species specific.

For example, the gradual disappearance of lamprey (ammocoetes) from annual surveys at Paprahia may be part of a cyclical or erratic migration pattern. Anecdotal cultural harvest reports from around the country suggests strong spawning runs may occur for this species in some rivers in some years and weak runs may occur in others. Although the life cycle of this species is still poorly understood (James 2008, Baker pers comm. 2016) it may be that spawning success of adult lamprey may only occur in some places in particular years. Ammocoetes are believed to remain in freshwater for 3-4 years post emergence from the egg (Kelso & Todd 1993, McDowall 1990). As a result, settled ammocoete larvae may be represented as part of the fish community over this period until that cohort metamorphoses to macrophthalmia before emerging from river sediments to leave the stream for the ocean. The decline of ammocoetes in Paprahia over the last 5 years to non-detection suggests that spawning in this stream over recent years has likely been very limited or has not occurred. Another possibility is that spawning has occurred but that the finer backwater sediments that ammocoetes typically inhabit had reduced in spatial extent within our sampled reach over time. This possibility is unlikely as backwater deposition areas were available and largely unchanged during the reporting period.

In contrast to lamprey, some observational evidence for torrentfish (a species which also declined to non-detection in this same stream) suggests that the number present at a site from which the species is known to occur may (at least in part) be controlled by the availability of fast water habitat present at the time of sampling. It is possible that the last 3 years (2011/12 - 2013/14) which have been unusually dry across the Waikato region, may have reduced the availability of fast water habitat for this species at this site, causing fish to drop out of the site in search of more preferred habitat.

Conversely other species at the same site such as redfin bullies which were consistently in very high abundance and bluegill bullies which were in consistently very low abundance showed no real change in spatial distribution within the same reach throughout the monitoring period. The spatial distribution and low number of bluegill bullies at this site is particularly interesting. Bluegill bullies are rarely detected within the Waikato region and we are yet to sample a site where more than 5 individual fish have been captured. In Paprahia we have consistently captured between 1 and 3 adult individuals from the same 30m section (2 adjacent subreaches) of stream for the last 6 years. These fish are typically captured in these sections along with redfin bullies, a species which is abundant throughout the entire 150m reach. Immediate questions that come to mind for the small bluegill bully population are; are these the same adult fish being recaptured each year? If they are not where is the recruitment coming from to support the small population? Why are they only in two adjacent subreaches? At this stage resources do not exist to specifically answer these questions but the regular detection of a small cryptic species in low abundance from the same stream segment over time provides some confidence that the methodology we are using is adequate for reach scale species detection and assessment. It is probable that a longer data record in conjunction with a regional and ideally national network of sites will enable closer examination of possible mechanisms driving population structure and function of particular species. Such data also illustrates the value of finer scale (sub reach information) for some species when undertaking these assessments.

One general issue worth noting for reference sites, particularly those with a high proportion of diadromous species is that while a selected site may exhibit minimal local anthropogenic effects, the site itself may not be immune from impacts that may be occurring at larger spatial scales. As discussed earlier, a proportion of annual recruitment of fish to support an existing reference site community may be originating from a number of other localities. Thus, site proximity to other relatively undisturbed catchments up and down the coast may be important for ensuring adequate recruitment/larval supply to support that local community. Effectively it could be argued that for fish assemblages possessing a high proportion of migratory species that classifying them as ‘reference’ or ‘minimally impaired’ may not be appropriate. Additionally, differing oceanic currents and coastal morphology (e.g. harbours v open coast) may favour retention or promote dispersal of larvae around coastlines (Fig. 21) which may conceivably influence recruitment success to particular catchments as suggested recently for torrentfish (Warburton 2016).

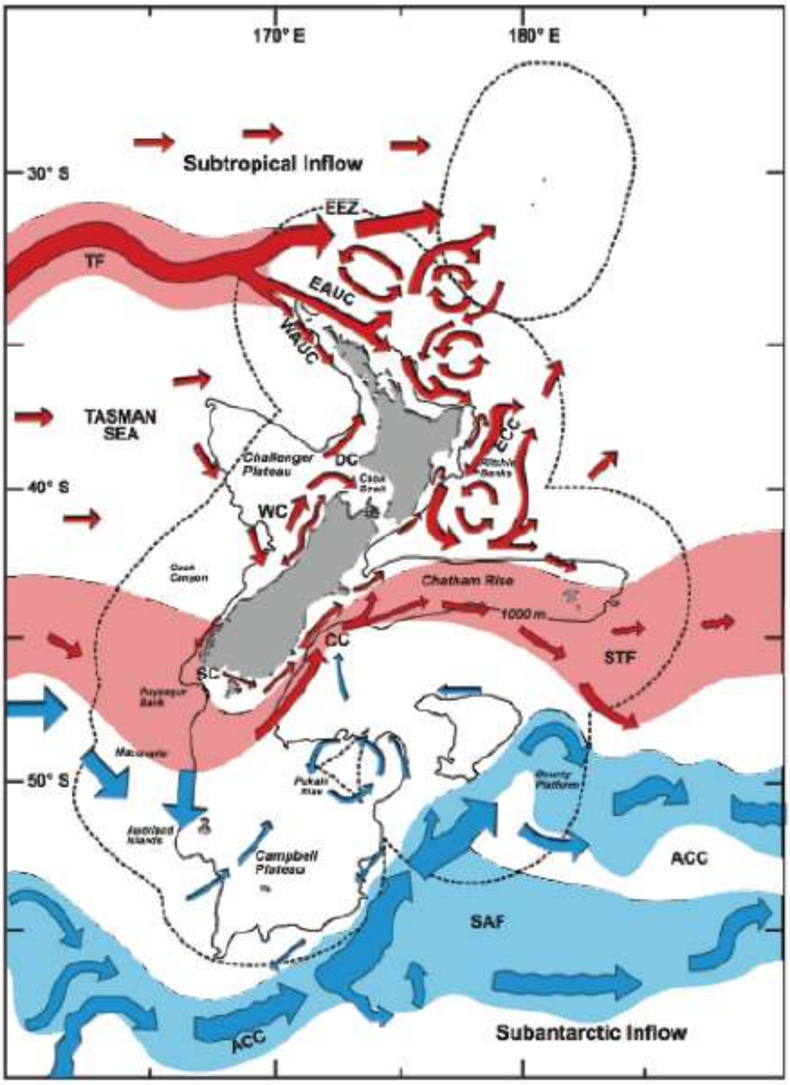


Figure 21: Map of major New Zealand marine currents. Adapted from Heath (1985) and Carter et al. (1998).

A well recognised phenomenon of the diadromous New Zealand fish fauna is that minimally impaired streams close to the coast tend to possess higher numbers and diversity of fish than streams further inland and at higher elevation (Leathwick et al. 2008). Theoretically then, effects of for example recruitment limitation may be more noticeable for minimally disturbed sites that are further inland where the likelihood of recruitment saturation (i.e. where species are controlled by density dependence) is potentially lower. While the WRC dataset is rapidly generating a range of hypotheses, it is likely that unraveling the mechanisms influencing migratory species distributions and recruitment (regionally and nationally) will require a collaborative and consistent effort of data collection from across the country. In the following section we discuss preliminary results to date for redfin bully, a species which at this stage appears to exhibit some interesting regional differences in population structure and where a national dataset could be called upon to assist this interpretation.

Potential emerging regional patterns – redfin bullies and eels

Redfin bully is a widely distributed endemic New Zealand species (McDowall 1990) which may be locally abundant, particularly in cobbly streams close to the coast. Although the species is known to penetrate more than 200km inland (NZFFDB), in the Waikato region this species is rarely found more than 50km from the coast. Widespread regional sampling indicates that a stronghold for this species in the Waikato Region is located on the Coromandel peninsula, particularly in streams draining off the eastern most ranges. In these streams redfin bully are consistently the most frequently encountered resident species in low to mid elevation (altitude) reaches. This species is also encountered in streams draining to the western coast of the region but some notable differences in population characteristics seem to be emerging between the coasts. For instance, in general and irrespective of whether a site is on developed land or reference, redfin bullies appear able to grow or live to a much larger maximum size in streams draining to the west of the region compared to the east.

Despite the generally larger size of west coast redfin bullies, over the last few years across every site where this species was present, a numerical decline at each site was recorded in association with limited recruitment. Across the same time period the opposite phenomenon seemed to be occurring on the east coast where numbers had increased and where a large proportion of the captured population comprised what we believed were young of the year (0+) fish. Recent collection of redfin bullies upstream and downstream of sampled sites has been undertaken to examine age-length relationships of this species between the two coasts and to confirm the length of 0+ fish. Extraction and examination of otoliths (ear stones) should provide useful insights into recruitment timing and possibly any differences in stream productivity (as related to fish growth and age) that may exist between eastern and western parts of the Waikato region.

Another major difference between the two coasts for this particular species is the number of sequential forested catchments between respective populations. Redfin bullies are typically closely associated with riparian canopy cover. The numerical stronghold for this species is the Coromandel peninsula where vast continuous tracts of conservation estate containing cobbly forested streams presumably supply large numbers of larval redfin bullies into oceanic waters off this coastline. In comparison, forest cover on the west coast is much lower and more fragmented. Differing landuse, possibly in combination with lower stream power in western drainages, tends to result in higher stream sediment loads and deposition of fine material. Fine sediment is known to negatively impact native fish species in New Zealand (e.g. Rowe et al. 2002; Ramezani et al. 2014; Greer et al. 2015; Harding & Jellyman 2015). For species like redfin bullies which typically utilize spaces beneath cobbles for egg laying and nest guarding, these general regional differences in land use and stream hydrology may provide some explanation for the lower relative abundance of redfin bullies in western parts of the Waikato region.

In addition to redfin bullies, eels were also encountered with sufficient regularity to initiate some regional analyses. Although both short and longfin eels were widely distributed throughout the sampled wadeable and reference network, some notable differences between the two species were evident. Most reference sites were dominated by longfin eels and in general whenever shortfins were captured at these sites, most of the individuals were elvers (<100mm). Interestingly relatively few longfin elvers were captured, either at reference sites or anywhere else in the random network.

Another interesting observation was that shortfin rather than longfin elvers were more likely to be detected at sites located at greater distances inland. In contrast, of the longfin elvers captured, most were detected in sites closer to the coast. So an obvious question is, why are there so few longfin elvers captured in the WRC fish monitoring programme? There are a number of possible explanations. Firstly it is possible that longfins less than 100 mm may behave differently to shortfin elvers at the same general life-stage (first 12-24 months in freshwater). For instance if longfin elvers staged for longer in areas closer to the coast or perhaps used deeper areas within the streambed, this could explain reduced catches relative to shortfins of similar size, and the tendency for capturing them closer to the coast. It is difficult however to postulate why this difference in behavior between the species would exist. Interestingly other researchers have also noted the absence of longfin elvers from typically longfin dominated streams in the South Island and this observation has prompted some exploratory experiments to evaluate the vertical depth positioning of longfin and shortfin elvers in porous baskets inserted into stream beds (Crow unpublished data). There is perhaps a weak suggestion that longfins may burrow deeper than shortfins but results are fairly inconclusive at this stage. Another possibility is that recruitment for longfins in more recent times is much lower than historically. Elver records at hydro dam traps around New Zealand certainly suggest that shortfin elvers are more common (numerically) than longfin elvers (all sites combined, Martin & Bowman 2016). Admittedly the proportion of short and longfin elvers differs greatly between sites and historical data indicates that recruitment in any given year can be highly variable for both species. To some degree lack of consistency in collected data and alterations to traps and their operational management at a number of sites has hampered any robust assessment over time (Haro et al. 2015).

Elvers aside, relatively few large eels were captured from wadeable sites, irrespective of whether sites were random or reference. Size structure information from the 144 sites sampled over 3 years indicates that most wadeable rivers only support low numbers of 'harvestable' eels. Preliminary data from netting sites where the sampled reach tended to be on average deeper (and less conducive to effective electrofishing) suggests that larger eels are likely to be more commonly encountered. Although our data are restricted to wadeable sites only, and that larger eels are probably more likely to be captured in larger rivers, there is concern among some groups and researchers that longfin eels in general have declined across New Zealand prompting a number of reviews and assessments (e.g. PCE 2013, Haro et al. 2015). Commercial catch records of longfins from the 1980's and 1990's tend to support this view but data are complex to interpret due to changes in the management of the fishery, accuracy of spatial and temporal fishing effort and changing market requirements (Haro et al. 2015). In more recent times it is possible that declines may have stabilised based on catch records but similar data interpretation issues persist.

Notwithstanding the national issues with data interpretation for eels, this programme and its rotational nature provides a platform for exploring relationships that may exist between various landuse gradients and fish population parameters for a wide range of species. Such relationships may then generate useful metrics for evaluating natural vs anthropogenic changes in the state of fish communities over time. Since sites are repeat monitored using the same methods over time, opportunities to evaluate site specific as well as regional changes are possible. For both regional and site specific assessments, it is expected that the analytical power and robustness of assessment of the data set will increase over time.

The current core set of reference sites provides a useful benchmark for evaluating how much inter-annual variation may exist for various metrics such as species diversity, QIBI, % balance of different species in an assemblage, absolute counts and relative abundance of individual and total species.

4.2 Potentially useful metrics

Random vs Reference fish 2011/2012 v 2014/2015

A number of potentially useful metrics to compare developed vs undeveloped regional sites appear to be emerging. In particular the QIBI score seems to exhibit promise as a key metric for differentiating sites with healthy functioning fish communities and those exhibiting signs of anthropogenic influence. Although the Index of Biological Integrity (IBI) approach is used extensively overseas (Karr 1981, 1987; Hughes et al. 1998) and has been demonstrated to perform well as a component for evaluating stream health, in New Zealand there has been some debate regarding its utility. One of the main criticisms for use of this approach in New Zealand is that the fish species richness is typically much lower than in many other overseas countries. Part of the argument is that with lower richness there is a fairly narrow range within the index for effectively grading stream health.

A further complication is that in New Zealand salmonids are an introduced fish that can impact on fish biodiversity but also have a requirement for good water quality. In effect for the purposes of the Waikato QIBI, salmonids have been considered as an 'honorary' native species (Joy 2007) reflecting good instream conditions even though they may be having a predatory or competitive influence on some local species and processes. Our experience and preliminary data for the Waikato region tends to suggest that salmonids are not exerting a measurable negative impact on native fish community diversity in the systems that we have sampled. There are likely to be a number of reasons for this. Firstly salmonids rarely dominate (numerically) any of the 1-4th order streams that we have sampled. Salmonids in the Waikato region tend to be better represented (numerically) in larger (non wadeable) rivers. Secondly, given the high levels of diadromy among the native fish fauna, regular annual recruitment is available from elsewhere to potentially offset predatory or competitive effects to some degree. Certainly the most clearly documented cases where salmonids have caused measurable numerical native fish declines have involved a range of non-migratory galaxiid species primarily in the South Island (Townsend & Crowl 1991. McIntosh 2000, McDowall 2006). In numerous cases complete extirpation of closely monitored non migratory populations in Otago, Canterbury and Southland has coincided with the invasion of salmonids (McDowall 2006) effectively raising their conservation threat ranking particularly over the last decade (Allibone et al. 2010, Goodman et al. 2014).

QIBI

Notwithstanding documented salmonid impacts in the South Island, until data suggest otherwise, inclusion of salmonids as 'honorary' native species in one component of the QIBI calculation for wadeable sites in the Waikato region seems acceptable at this point in time. The use of QIBI appears to have promise as a metric for evaluating regional anthropogenic impacts on fish populations. So far in our programme scores have been significantly higher at reference sites (QIBI median = 56 in 2011/2012 and 2014/2015 and the variance much lower (at a regional level) than those calculated for random sites (QIBI median = 32 both years). The QIBI appears to work particularly well in degraded lowland wadeable sites in the Waikato where a host of tolerant non migratory invasive fish such as brown bullhead catfish, gambusia and goldfish are present.

Extent estimates

In this report we also used the probabilistic network design to report on extent estimates for the QIBI metric. In short it is apparent that the mean and median fish QIBI scores are much lower and more variable in streams running through developed land. Using data from the 144 random sampled sites over 3 years it is possible to extrapolate likely QIBI scores throughout the entire wadeable Waikato region river network (see Collier and Hamer 2012). Based on fish QIBI data, it was estimated that around 21% of the region's wadeable river network on developed land would support fish communities with an 'excellent' QIBI score (c. 2100 network km) while more than half of the network is predicted to exhibit 'moderate- poor' QIBI scores (c. 4560 network km).

The likely mechanisms influencing IBI scores in river reaches on developed land include reduced water quantity and quality, loss of instream cover (e.g. Baillie et al. 2013) and loss of or impaired river connectivity. Additionally reduced densities of drifting terrestrial invertebrates from reduced surrounding forest vegetation, and increased siltation and erosion which can lead to loss of pools and interstitial space (Leprieur et al. 2006; Eikaas et al. 2005 a,b; Ramezani et al. 2014).

Alteration in land use, particularly the conversion of forestry and native bush to pasture, appears to be a major contributor to some of the observed declines in freshwater fish nationally, including the Waikato region (Joy 2009). Increasingly, land management that increases fine sediment inputs to streams is being recognised as one of the most important stressors impacting aquatic life in New Zealand (e.g. Rowe et al. 1999, Richardson & Jowett 2002, Rowe et al. 2002; Ramezani et al. 2014; Greer et al. 2015; Harding & Jellyman 2015).

Future metric development

At the time of writing (2015/2016), the fish SOE programme is currently into the re-sampling of the second set of 3 year rotation random sites. By the end of 2016/2017, the entire set of random sites (144 random sites) will have been re-sampled twice. Information collected through this programme has been designed to enable evaluation of fish communities and community attributes across a wide range of levels. For instance, over time it will be possible to explore the 'state' of individual species within our region. In examining individual species it would be possible to assess their frequency of occurrence at sampled sites over time and any temporal recruitment trends for certain species at both the site and regional scale. Other useful analyses would be to examine any changing trends in the overall populations themselves such as the proportional representation of species and size classes within fish assemblages over time, whether invasive species are becoming more widespread in terms of numbers or relative abundance across the region or whether overall the native and/or exotic diversity of fish is increasing or decreasing. Again opportunities to assess these fish community aspects and parameters would be possible at both a site and regional scale. At the regional scale it will be increasingly possible to use extent estimates for individual species to for instance look at the number of river km over time that individual species and communities are likely to be occupying.

An additional exciting prospect will be to merge WRC's fish and invertebrate programmes together to explore the potential to develop a more holistic and integrated ecological assessment of wadeable river health for the Waikato region. Since the invertebrate programme collects various other data within the same reaches, in the same season (e.g. information on physical stream and riparian habitat, periphyton, aquatic plants etc), significant opportunities to isolate and or integrate these components into an overall river health score exist at both the site and regional scale.

A long term goal would be to automate many key analyses and potentially to establish key breakpoints/thresholds for various parameters or combination of parameters influencing particular species and or aquatic communities.

4.3 Other aspects for consideration

EFM vs Netting – consistency in metrics from these methods

It is apparent that the fishing method used at a site can potentially produce differing results with respect to various fish population characteristics (i.e. netting v electrofishing). For instance the likelihood of capturing a greater proportion of large eels seems to increase when using netting protocols compared to electrofishing. It is also possible that other community attributes may differ between these two protocols. Such differences are not overly surprising considering that one approach is passive (netting) and the other active (electrofishing). Different fish may also be more or less conducive to being captured by the differing approaches. For instance it is likely that electrofishing is much more effective at detecting elvers at a site as the machine can actively extract these fish from within river bed substrates than netting. Alternatively nets in deeper, slower habitats will probably be more effective at catching larger eels and other species than electrofishing where the machines capabilities may be limited by depth and flow. While it is important to recognize these potential biases between the protocols (particularly when interpreting results), the differences are not necessarily problematic for regional fish community state and trend assessment. Of specific importance in this regard is consistency of approach for enabling a relative site specific or regional assessment. In other words providing the same set of sites are evaluated using the same gear type and effort between monitoring periods it should be possible to evaluate site specific and regional changes to fish communities.

For some analyses in this report it was necessary to exclude a small number of sites from the random network to ensure that the same set of sites and protocols were being evaluated between the 3 year rotation period. For instance a number of sites sampled in 2011/2012 were dry in 2014/2015 and for a small number of other sites landowner access to revisit and resample (3 years later) was refused. Effectively it was necessary to exclude these sites from assessment to ensure comparability of the same sites between the rotation period.

It is conceivable that the number of sites for long term rotation comparisons may slowly reduce over time so it is important that a backup list of sites exists to replace those that may need to be dropped to ensure sufficient spatial site coverage is maintained within the programme.

4.4 Conclusion

This preliminary assessment of WRC’s fish monitoring programme has demonstrated that it is possible to consistently sample freshwater fish communities using standardised national protocols at a site and regional scale. In general, physical, chemical and biological parameters were consistent and exhibited minimal variation at individual reference sites between years. These protocols have been used across a large number and diverse range of wadeable river sites selected at random within the Waikato region. Consequently we consider that a robust baseline of information on fish communities in wadeable streams within the Waikato now exists to monitor their ‘state’ (and any changes over time) to better inform policy development and management decisions. The QIBI at this stage appears to offer promise as a useable metric that could be incorporated with other physical chemical and biological metrics to better evaluate wadeable stream health. Other metrics such as the ratio of native to exotic species and trends in distribution and extent of individual species may also show promise for assessing ‘state’. For individual species and site specific assemblages, other more detailed aspects such as the size structure or proportional balance of the assemblage over time warrant closer evaluation as more data are collected. Such data may provide more site specific as opposed to regional assessment of potential pressure(s). It is highly likely that greater national adoption of these standardised protocols and systems (where none other currently exist) provide a unique opportunity for robust national reporting on freshwater fish communities. Future proposed amalgamation of data from the REMS invertebrate and fish monitoring programmes and development of associated metrics is likely to provide a much more comprehensive and holistic assessment of the state and health of the Waikato wadeable river network.

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Appendix 1

Additional parameters measured at all random and reference sites for future potential inclusion in developing a combined river health score.

Parameter	Protocol used	Output
Water quality (1)	Single 1L water sample	N,P, Ecoli, values
Water quality (2)	YSI 85 handheld, one off	Dissolved oxygen, conductivity, temperature
Canopy cover	Densiometer	Shade score
Physical Habitat	WRC stream habitat assessment sheets. Collier & Kelly 2005	Habitat score
Invertebrate communities	Soft and Hard Bottom protocols. Stark et al. 2001	MCI score
Periphyton	Collier et al. 2014	Periphyton enrichment, channel clogginess
Substrate	Collier & Kelly 2005	% substrate composition

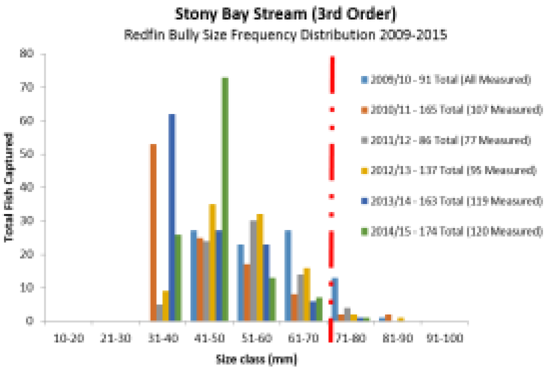
Appendix 2

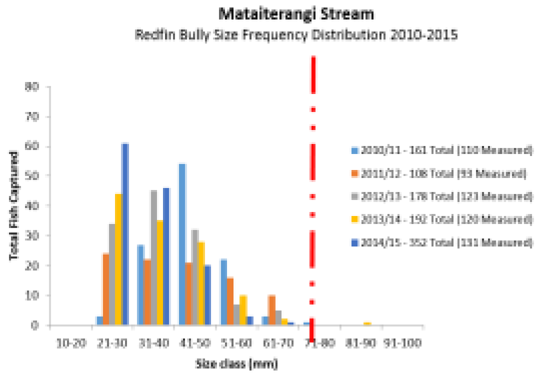
Bed disturbing flows used to determine sampling stand-down period during seasonal SOE monitoring programme.

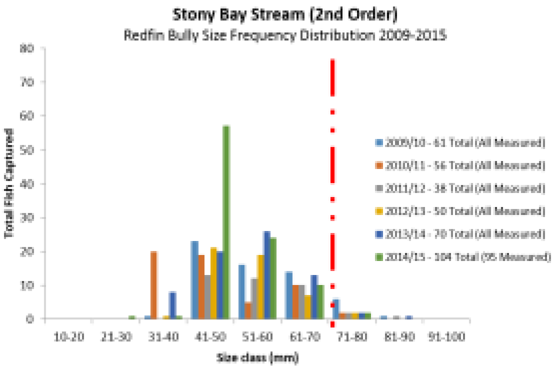
River	Site	Catchment Area	BED DISTURBING FLOW (m3s-1)
Awakino	Gorge	226.0	100
Waipa	Otewa	317.0	90
Mangaokewa	Te Kuiti PS	173.2	60
Mangatangi	SH2	196.0	30
Mangawara	Jefferis	98.0	20
Ohinemuri	Karangahake	287.5	133
Kauaeranga	Smiths	122.0	80
Tairua	Broken Hills	117.0	70
Tauranga/Taupo	Te Kono	199.5	75
Whakapipi	SH22	44.4	10
Piako	Kiwitahi	105.0	15

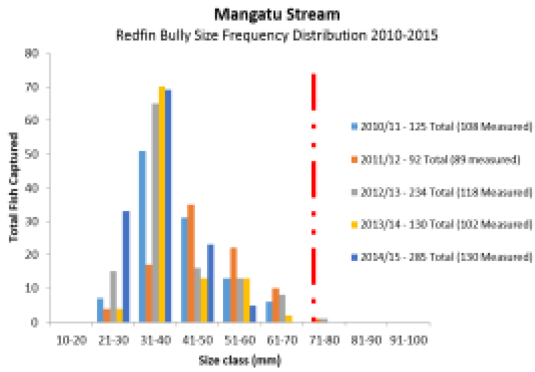
Appendix 3

Redfin bully east coast reference sites – note upper fish size differences between east and west (denoted by red line >70mm).



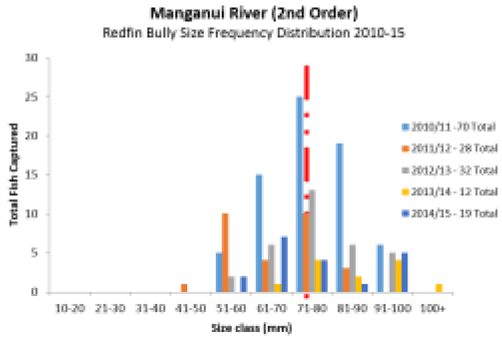


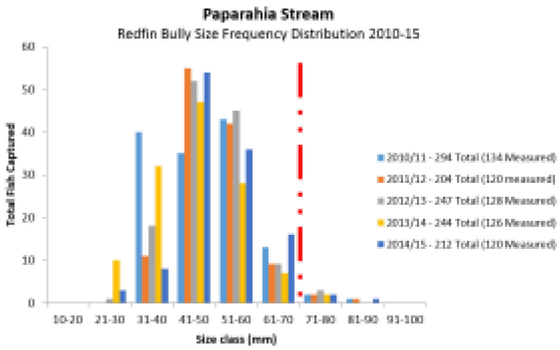


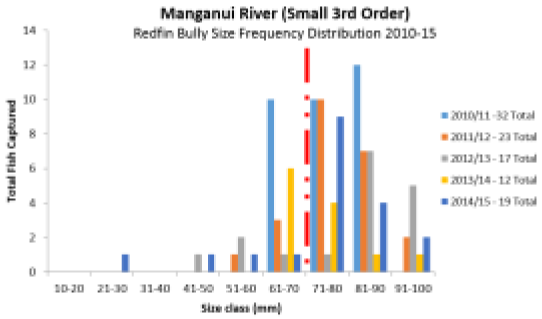


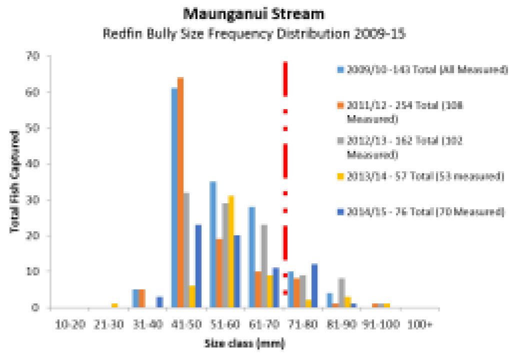
Appendix 4

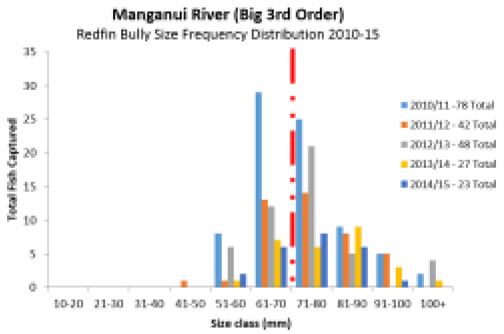
Redfin bully west coast reference sites – note upper fish size differences between east and west (denoted by red line >70mm).





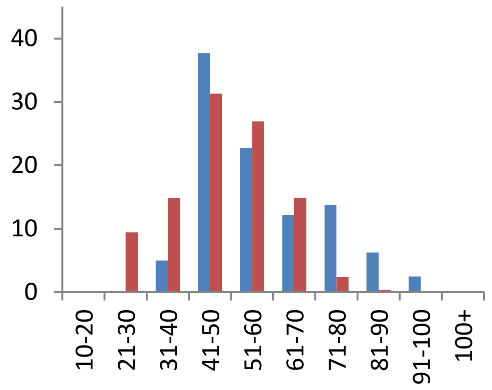




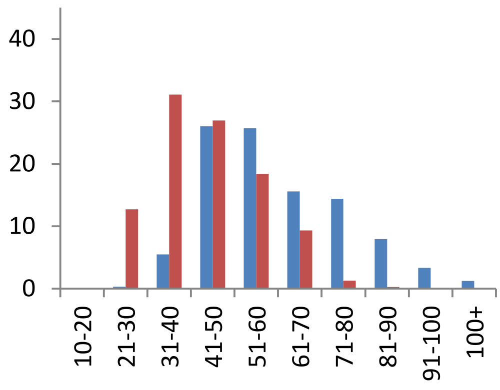


Appendix 5

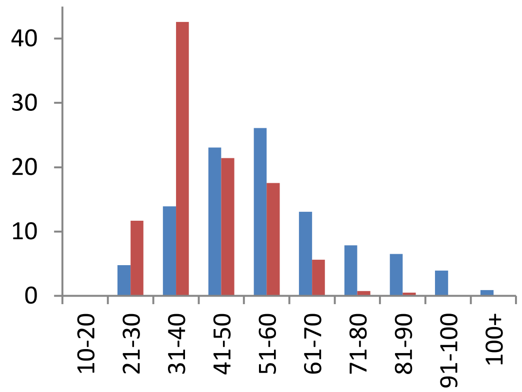
Mean proportion of redfin bully by size class at annually fished reference sites draining to east (red) v west (blue) x year (indicated below figure)



2011/2012 E n=4 sites, 297 fish, W n=5 sites, 321 fish)



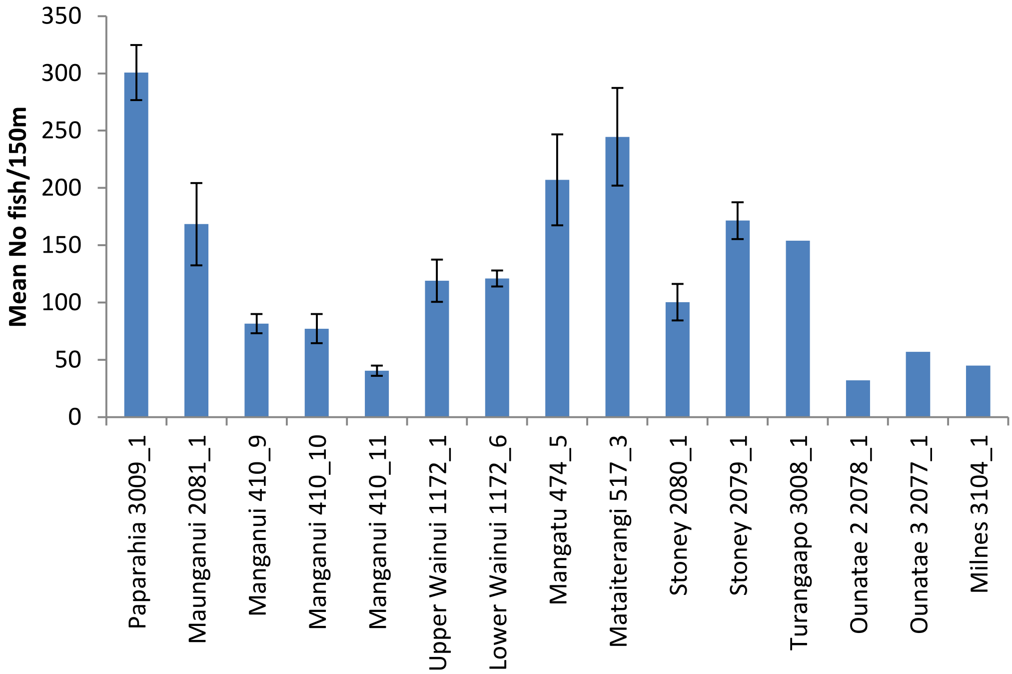
2012/2013 E n=4 sites, 386 fish, W n=5sites, 327 fish



2013/2014 E n=4 sites, 411 fish, W n=5 sites, 230 fish)

Appendix 6

Mean number of fish/150 m. See Fig 1. A) for number of years on which average is based for each site.



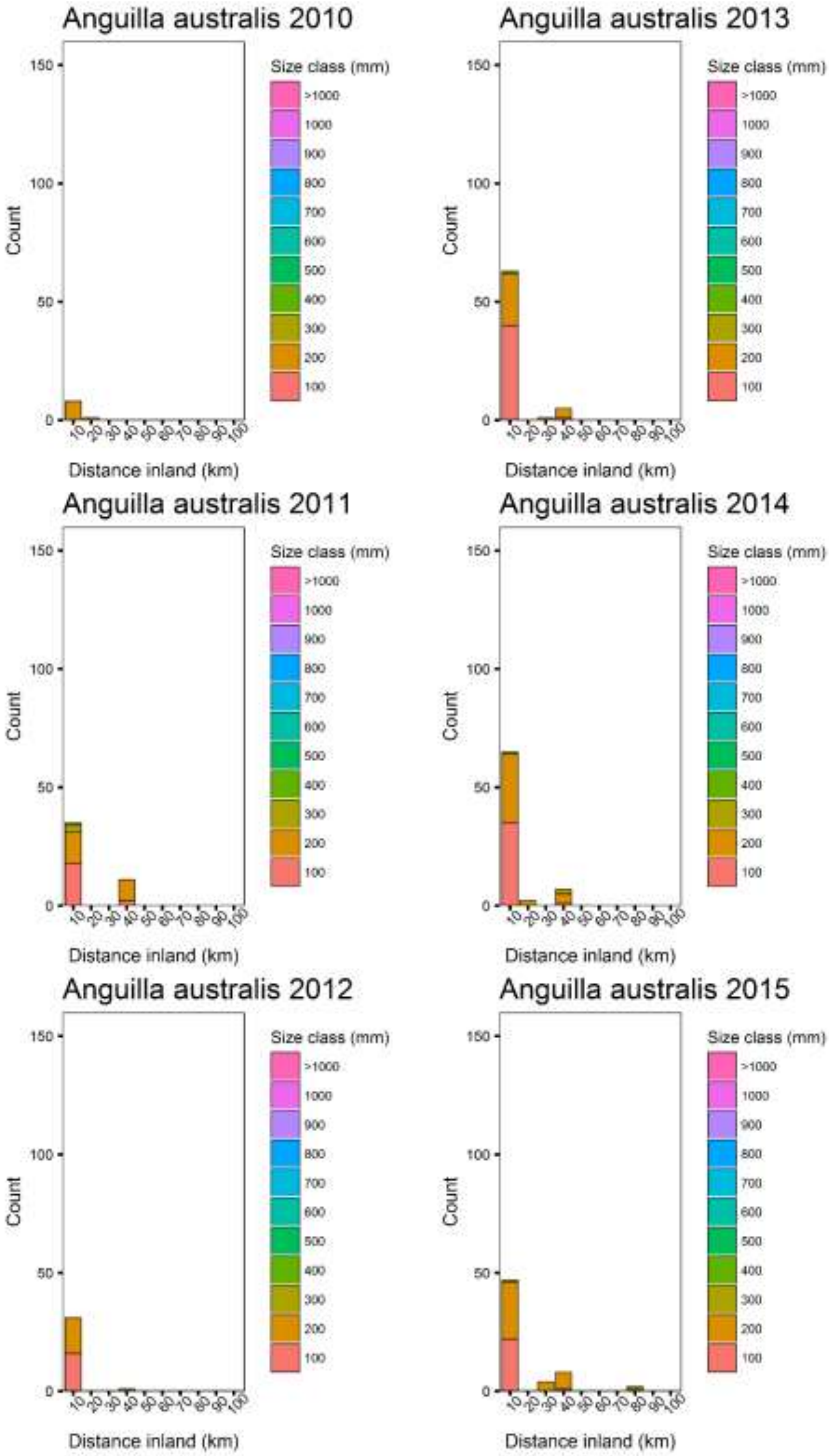
Appendix 7

QIBI Extent estimates for all random electrofishing (EFM) and trapping sites sampled within the Waikato Regional Council fish SOE monitoring programme

EFM (123 sites)	IBI category	Est % network	StdError.P	Est network km	StdError
	Excellent	21.54	3.39	1925.95	315.96
	Good	17.52	3.03	1566.50	267.36
	Moderate	28.23	3.92	2523.47	363.00
	Poor	16.70	3.47	1493.12	327.00
	No Fish	16.00	3.35	1430.24	323.93
Trapped (21 sites)	Excellent	17.76	6.76	183.87	61.70
	Good	29.78	9.17	308.36	90.69
	Moderate	33.15	10.95	343.28	109.70
	Poor	19.32	11.64	200.06	140.17

Appendix 8

Number of shortfin (top) and longfin (bottom) eels of different sizeclass by distance inland from all reference sites sampled from 2010 – 2015.



Number of longfin eels of different size class by distance inland from all reference sites sampled from 2010 – 2015.