Eddleston Water restoration project: Fish monitoring programme 2017 - 2019

The Clubhouse, 106 Biggar Road, Edinburgh, EH10 7DU
t: 0131 445 1527 e: trust@forthriverstrust.org

forthriverstrust.org forthriverstrust @ForthRiversT ForthRiversTrust

Report prepared for Tweed Forum

April 2020

Quality management

Author(s):	Tommy McDermott (On behalf of Forth Rivers Trust James Hunt (Tweed Foundation) Joanna Girvan (Forth Rivers Trust) Colin Adams (University of Glasgow)
Surveyor(s):	James Hunt (The Tweed Foundation) Kenny Galt (Tweed Foundation) Tommy McDermott (On behalf of Forth Rivers Trust) Patrick Barbour (The Tweed Foundation) Joanna Girvan (Forth Rivers Trust) Sylvian Barry (Forth Rivers Trust) Iain Reid (Forth Rivers Trust) David Eastwood (Forth Rivers Trust) Jonathan Louis (Forth Rivers Trust) Jack Wootton (Forth Rivers Trust) Mike Voermans (Forth Rivers Trust)
Data analysis/GIS:	Tommy McDermott (On behalf of Forth Rivers Trust) Colin Adams (University of Glasgow) Jennifer Dodd (Edinburgh Napier University) Mike Voermans (Wageningen University)
Date:	April 2020
Project Number/Document Reference:	P2017004
Client:	Tweed Forum

Executive Summary

In 2017, Forth Rivers Trust[1] was contracted by Tweed Forum to develop and execute a three-year monitoring programme investigating the response of native fish species to river restoration efforts on the Eddleston water. This restoration project involved several significant works, and of relevance to this summary was the channel re-meandering carried out at Cringletie, Lakewood and Shiphorns.

This monitoring programme was delivered in collaboration with The Tweed Foundation and The University of Glasgow. The contract involved the following key tasks:

1. A channel walkover to understand the habitat diversity and distribution on the restored and unrestored channels, and to identify suitable electrofishing sampling locations;
2. An *a priori* power analysis to establish likely sample effort necessary to determine differences (if any) between fish population measurements at the sampling locations;
3. Fully quantitative electrofishing at the identified sites;
4. Spawning redd surveys; and
5. Related analysis, interpretation and reporting.

A habitat walkover was carried out in July 2017 from Cringletie to Shiplaw. This walkover identified potential electrofishing sites and described and quantified the available habitat in areas of restored and unrestored channel. Power analysis identified the potential for a grouped Control-Treatment based sampling design to identify large differences in fish density between restored and unrestored sites.

Fully quantitative electrofishing sampling based on Scottish Fisheries Co-ordination Centre protocols was carried out at the 12 sites chosen for each of the three years (2017, 2018 and 2019). Fish were processed in standard fashion; retained from each run (of three) until sampling was complete. They were then identified to species (or genus for *Petromyzon* lamprey juveniles), measured and the first 50 salmonids weighed, prior to return. Habitat data was recorded as per the SFCC recording form.

Redd surveys were carried out using a method deployed in Scotland by Forth Rivers Trust, developed from the American Salmonid Field Protocols handbook. The channel was walked during the spawning season (November – December) and redd presence, along with associated descriptive data, was recorded.

Two assumptions were made prior to data analysis. Although the programme desired to establish the effects of restoration on all species, the SFCC approach is prioritised towards salmonids. Therefore, other species were removed from further consideration due to low numbers and a likely unrepresentative catch. Secondly, salmon on the Tweed often smoltify after their first winter, while in general salmonid juveniles displace to different parts of the catchment for their first winter. Therefore, only data on salmon and trout fry (juvenile fish hatched during the survey year) were used in analysis.

All data were tested for normality prior to analysis and either transformed as required or suitable non-parametric tests identified.

Mixed effects modelling was used to establish differences between control and treatment locations for salmonid fry density, length and length variance. Habitat data, comprising over 77 potential parameters was compressed using visual review to remove redundant parameters (defined as parameters with the same value for all sites) followed by

[1] Formerly River Forth Fisheries Trust

Principal Component Analysis. Identified principal components were then tested for differences in habitats between control and treatment sites, and if these habitat differences had any relationship with salmonids.

Spawning activity data was grouped into control or treatment categories. These were then standardised to account for differences in channel length for each category. Linear modelling (or non-parametric alternatives) was used to establish any differences in spawning activity between control and treatment areas.

The results showed no difference per 100m^2 in the numbers of trout and salmon fry between the restored and unrestored sites surveyed. There was, however, a strong difference between years for Atlantic salmon fry, with 2018 numbers lower than the other survey years.

Atlantic salmon fry length was not dependent on year or restored/unrestored sites; however, variance in length was lower in 2018. Trout fry were smaller in 2018 when compared with 2017 and 2019, although there was no relationship between length and restored or unrestored areas for that species either.

PCA grouped the habitat data into six important components (accounting for 65% of the total habitat dataset variation). There was a significant difference in the first habitat component scores (33% percent of total variation explained) between restored and unrestored areas. This demonstrated that control (unrestored) sites were more strongly associated with an increased degree of complex bank vegetation, coarse substrate and energetic flows - and a decreased degree of macrophyte cover and simple bank vegetation, than restored sites. There was no relationship between any PCA components and salmon and trout data.

There was no relationship between relative redd counts and control or treatment reaches; however, there was once again a significant relationship with sampling year, with relative redd (redds/km) abundance decreasing year on year throughout the study period. This observation was caveated heavily however.

The results of the sampling programme show that all sampled sites from restored areas are now as productive as unrestored sites (i.e. complete recovery from channel construction) as there is no significant difference between the densities and lengths of salmon and trout fry. However, using density as a measure does not consider the relative abundance of habitat. Restored lengths of channel have an area that is least 10% greater than the area of the original unrestored channel, therefore, based on the absence of a difference between restored and unrestored sample locations throughout the study period, it is possible that the restored sections may now produce more fry than if they had remained unrestored. However, the Eddleston was not an unproductive river for salmonids prior to restoration.

Despite the above there was an opinion among surveyors that the new sections of channel felt underdeveloped. The scarcity of bank undercuts and woody debris features coupled with the immaturity of riparian trees means that cover is not yet fully developed in restored sections, leading to the possibility that maximum fish abundance here is yet to be realised.

Interannual variation is a strong signal emerging from the data and was probably most notable for redd counts with a year on year decline. However, collecting robust redd data is difficult and related to factors such as general flow patterns, spawning timing, water clarity and on-the-day light conditions. Electrofishing data from 2018 was related to significant reductions in: density (of salmon fry), length (of trout) and length variance (of salmon fry). The lower salmon fry length variance could be related to the lower count of fry, with the reduced density caused by very cold periods during March and April 2018, low flows in autumn 2017, or a combination of both. The smaller size of trout fry may again have been related to the very cold mid-spring period in 2018. The results highlight the often-overwhelming role played by interannual factors in salmonid survival and production which potentially can mask other factors such as improvements in habitat.

The limitations of the methods were exposed, particularly electrofishing. While salmonids are a key component of Scottish aquatic ecosystems, other species such as eel, lamprey, stickleback and minnow are not confidently enumerated using the standardised electrofishing approach. There may be a future requirement for a restoration-specific fish monitoring protocol to be developed.

Contents

1. Introduction

1.1. Background

The Eddleston Water is part of the River Tweed catchment, which hosts significant populations of Atlantic salmon (*Salmo salar*) and is a Special Area of Conservation (SAC) with salmon as a qualifying feature. In addition, brown trout (*Salmo trutta*, and includes the anadromous sea trout), European eel (*Anguilla anguilla*) and three species of lamprey (sea, river and brook; *Petromyzon marinus*, *Lampetra fluviatilis* and *L. planeri*), common minnow (*Phoxinus phoxinus*) and stickleback (*Gasterosteus aculeatus)* are among the native species found in the catchment. Grayling (*Thymallus thymallus*) and stone loach (*Barbatula barbatula)* are also present, but not native to the river.

The Eddleston Water rises in the hills to the south of Edinburgh, flowing to Peebles where it joins the Tweed. It drains a catchment of 69km^2, mostly used for agriculture. The Eddleston has undergone significant morphological alteration due to land use and infrastructure development, peaking between 1770 and 1815 (Harrison, 2012). This has resulted in a straightened channel fixed in position for potentially several centuries, embanked and disconnected from its floodplain, which may have caused a decline in certain aquatic ecological communities. These changes will have altered the hydromorphological processes on the river with its straightened channel conveying flows rapidly downstream. A map of relevant sections is presented overleaf in Figure 1.1.

A report into the feasibility of restoring the Eddleston to a more natural form was produced by Werrity et al. (2010) along with a proposed strategy to deliver the works. An implementation programme was subsequently launched with actions delivered by the Tweed Forum, funded by the Scottish Government (Table 1.1).

Table 1.1 Timeline of interventions.[2]

Site	Intervention	Year
Shiplaw	Livestock exclusion/ tree planting	2012/2013
Longcote	Livestock exclusion/ tree planting	2012/2013
Cringletie	Remeander	2013
Lakewood	Remeander	2013
Shiphorns	Remeander	2014
Milkieston	Cringletie-Shiphorns connection	2016

On the Shiplaw and Longcote tributaries interventions were relatively light touch, with the creation of grazing free zones and tree planting. The channel remeandering involved significant works with a design process and the creation of new channels with older channel sections dewatered and closed off.

Natural Flood Management was a key driver of the works; however, improvements to ecological functioning, biological community structure, diversity and abundance were part of several other desired outcomes. Numerous other teams are currently reporting on fields as diverse as macrophytes, benthic invertebrates, channel geomorphology and catchment hydrology, with reporting for those receptors to sit alongside this report. It is then planned that ecological,

[2] Further information on the measures implemented on the Eddleston catchment are available on the Tweed Forum website.

geomorphological and hydrological data will be brought together in a final study to assess the linkages between these related and co-dependant receptors.

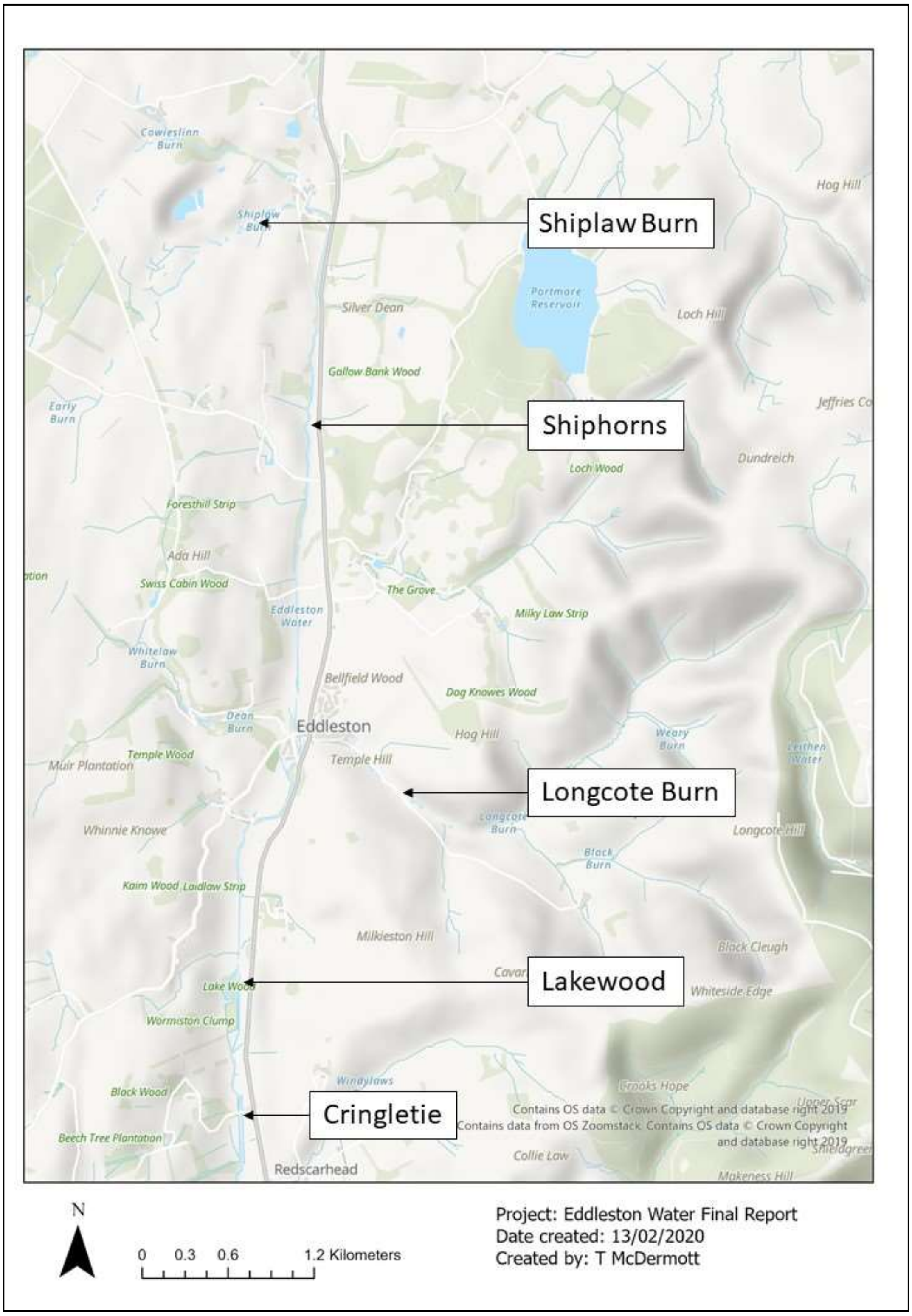


Figure 1.1. Outline of relevant mainstem channel areas and tributaries

1.2. Fish and restoration

River restoration is a catch-all term incorporating everything from simply increasing in-channel cover to reconnecting rivers to former routes or the creation of entirely new channels. The literature does provide a synthesis of fish responses, but it should be remembered that these responses are often in relation to very different types of schemes that are included within the general restoration field.

River Restoration is now commonly used to return rivers to a more functional natural state, and, experience with stream restoration for the enhancement of salmonids suggests that, when projects are implemented correctly, sizeable increases in fish production can be achieved (Roni et al., 2010). It is presumed that habitat restoration will improve fish communities by simply creating more functional habitat, but this simplistic view may not be accurate (Palmer et al., 2009), and may fail to consider the complex inter- and intra-specifics of salmonid life history dynamics (Einum et al., 2008). Therefore, making direct links between channel restoration improvements in fish communities is difficult (Roni et al., 2008), and poor understanding of fish stocks before restoration, limited scope and poor monitoring can make it difficult to fully assess the benefits.

The effectiveness of restoration to improve fish populations was reviewed by Kemp (2010). Improvements varied enormously and were influenced by many other factors (Armstrong et al., 2003); life-stage and seasonal changes in habitat use (Heggenes & Gunnar Dock, 2001), the size of the rivers restored (Miller et al., 2008), and the relatively small scale of the interventions (e.g. Roni et al., 2002). As a result, many programmes report inconclusive results on fish populations or may appear to have failed to demonstrate any improvement. In a review of over thirty restoration measures, Marttila et al. (2019), found an overall positive influence of restoration on salmonid fry density, but these benefits often fell short of the desired outcomes.

Several issues around identifying the benefits of river restoration exist. These include unrealistic expectations of the scale of impact restoration can have on fish communities, and inappropriate monitoring programmes are common (Liermann & Roni, 2008). Some examples do exist where positive responses in salmonid production have resulted from habitat intervention. Nislow et al. (1999) have shown that the introduction of instream structures to increase the abundance of preferred habitats led to increased retention of juvenile Atlantic salmon during the critical first summer period; Gargan et al., (2002) reported improvements to the populations of Atlantic salmon parr in reaches following creation of in-stream structures to restore substrate and flow complexity; Jong et al., (1999) showed a positive response in Atlantic salmon parr abundance in response to instream structures placed into Newfoundland streams, attributing increases to improved habitat complexity reducing competition; and Solazzi et al. (2000) provided evidence of a doubling of smolt production in Coho salmon in the four years following establishment of in-stream structures to improve habitat quality in small catchments (<20km^2) in the Pacific northwest. However, these responses were often due to restoration involving complex instream habitat manipulations, which are not a common feature of the Eddleston restoration project.

In Scotland, our understanding of the response of fish communities to river restoration is impeded by the low number of significant restoration interventions coupled with a history of poor or absent monitoring of fish stocks specifically in response to restoration. On the Spey, there may be some data on fish densities potentially responding to channel remeandering - salmon fry densities are higher and on a distinct upward trajectory in areas of the river which were restored when compared with a control location. However, this pattern requires further investigation and validation with further years' data (Spey Foundation, unpublished). On the Nith, restoration signals have been complicated by stocking; however, it is possible that habitat improvement has been the paramount influence on population recovery (D. Parke, pers. comm).

1.3. This report

This document represents the results from a three-year monitoring study to assess the potential impact of the Eddleston restoration on fish populations. It aimed to answer key questions including:

I. Does restoration alter the abundance of fish in restored areas versus unrestored areas;
II. Does restoration alter the size of individual fish in restored areas versus unrestored areas;
III. Does restoration alter spawning effort in restored versus unrestored areas; and
IV. Does restoration alter habitat between restored and unrestored areas and are any observed differences related to fish communities.

To achieve this, several different methods were brought together to design and deliver a tailored monitoring programme. These were:

I. Monitoring programme design using Power Analysis and pre-survey, walkover-based site identification.
II. Mainstem electrofishing over three years; and
III. Redd surveys over three years.

It was hoped to investigate these questions for all native species; however methodological constraints (see Section 2) meant that, in reality, only Atlantic salmon and trout could be assessed.

An additional contract element, to continue semi-quantitative monitoring of fish communities on two Eddleston tributaries was also carried out, and is presented in Appendix B and C.

1.4. A note on salmonid life history and wording

The life history of salmonids is complex, with multiple stages defined by optimum habitat niches. Here, salmonid refers to Atlantic salmon and brown trout (including its anadromous form, sea trout).

Adult fish spawn in redds during winter. Approximately three months later these eggs hatch into alevins that remain within the redd area before “swimming up” into the water column as fry (within one to three months depending on environmental characteristics). Fry are often termed 0+ juveniles. In the lead up to the first winter these fry take on characteristic “thumbprints” along their flanks after which they are known as parr. Parr can remain in the river for 1-3 (generally) winters prior to either smolting and migrating to sea (salmon and sea trout) or entering a subadult phase before maturity (brown trout). They are often denoted as >0+ (i.e. older than fry) or, if they have been aged, the more specific 1+, 2+ or 3+ juvenile nomenclature can be used. In this report juvenile salmon and trout are termed either as fry or 0+ juveniles, and parr or >0+ juveniles. Due to survey timings, no smolts were caught, and where large brown trout were recorded, they were assigned as adult based on the expert judgement of the survey team supported by formal scale-based aging where necessary. Adult trout were not used in any data sets due to their very small number and preference for deeper water which wasn’t sampled.

2. Methodology

2.1. Monitoring programme design

Historic data assessment

To facilitate an understanding of pre-restoration fish populations, an assessment of available historic data was carried out. This dataset was provided to the project by the Tweed Foundation, and site codes, location and sample availability are presented in Table 2.1 and presented overleaf in Figure 2.2. The method used to collect this data is the same as described in Section 2.2.

Table 2.1 Background biological data availability

Code	Type	Easting	Northing	Years available (one sample per year)
ED01	Efishing	324283	648995	1988,1995,2000,2003,2006
ED00B	Efishing	323021	649971	1988
ED02	Efishing	323793	645604	2000,2003,2006
ED02C	Efishing	324375	642575	1988
ED03	Efishing	324789	641655	1995,2000,2003,2006,2015,2016

The pooled average density from each site for salmon and trout is shown below in Figure 2.1. Error bars represent standard error.

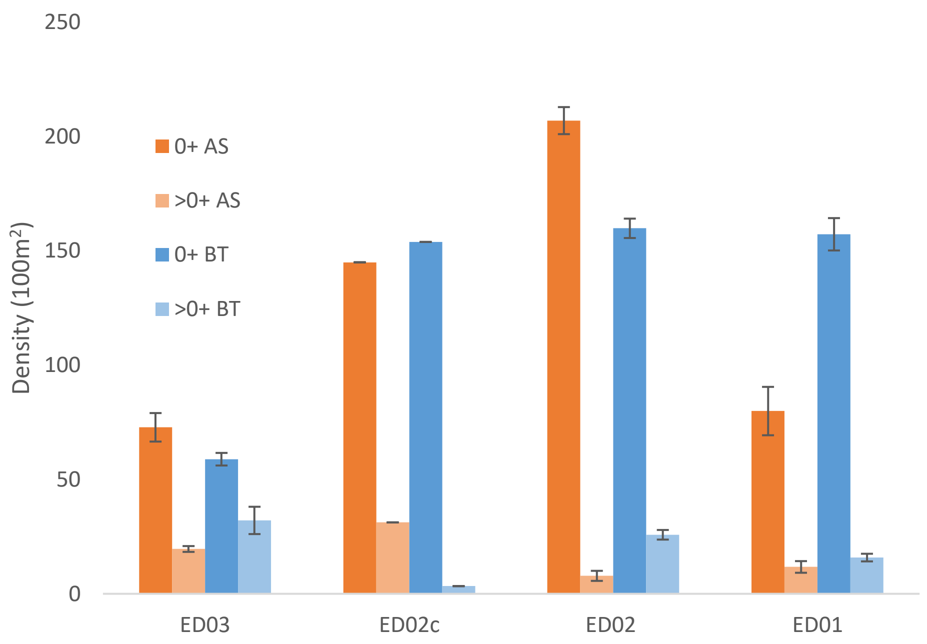


Figure 2.1 Densities of salmonid fish (per 100m) at survey locations on the Eddleston prior to restoration

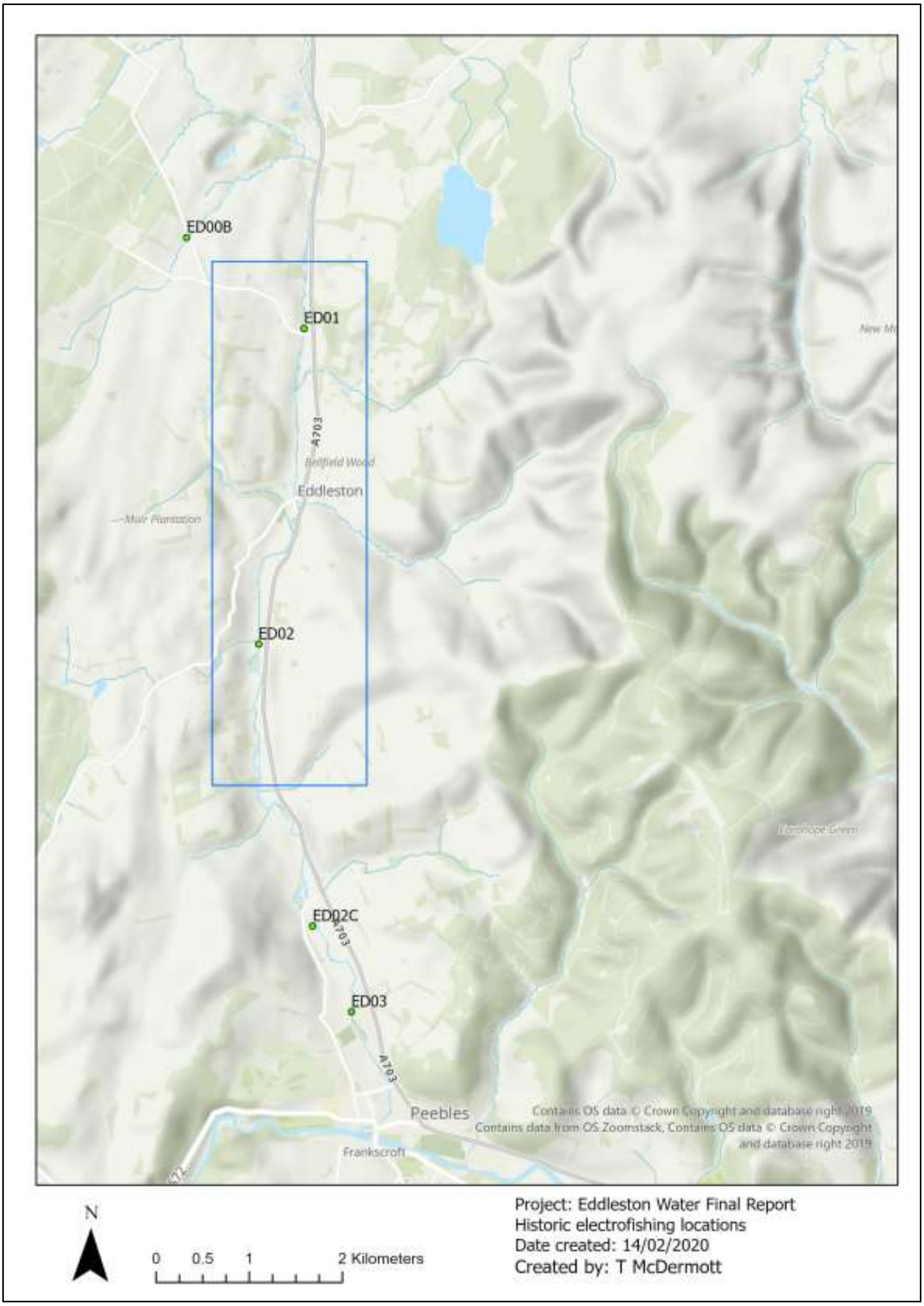


Figure 2.2. Location of historic data with current experimental area in blue

Although the method used is consistent, sampling effort within this dataset (i.e. number of samples per site, and time series length at each site) is inconsistent. However, the data do suggest that the Eddleston pre-restoration was (and quite likely remains) a productive channel for salmon (juveniles) and trout. During the most recent SAC assessment (RAFTS, 2014), 66% of the Tweed sites were categorised in the upper quintile for both 0+ and >0+ salmon, and the values presented in Table 2.2 are consistent with those from the fully quantified samples within that Condition Assessment.

While this dataset is useful for understanding potential population sizes, sample locations are often distinct from the restored areas, insufficient in sample effort, or too old to accurately describe a baseline condition that is confidently relevant to current fish populations on the Eddleston.

Monitoring Framework

Two potential monitoring frameworks were proposed at programme start-up, Control (unrestored) - Treatment (restored) and Before-After-Control-Impact (BACI, Stewart-Oaten et al., 1986) designs. The main difference between the two frameworks is that BACI provides a much more robust temporal measure of change, as any site under investigation will have been surveyed appropriately prior to a change being introduced. Although the BACI design potentially supports the optimum level of investigation, several factors resulted in a preference for the Control-Treatment approach.

The first is that there is technically no historical 'before' state for the restored channel, as these reaches are new channels. Secondly, although some data from the Eddleston catchment prior to restoration is available (See Section 2.2), it has not been collected in a consistent fashion and the results from each site may reflect a more or less productive phase for salmon and trout in the Tweed relative to this study period. Subsequently the historic data may not provide suitably robust baselines.

Sample size identification and site location

Two approaches were combined to inform the appropriate number and location of the sample sites; Power Analysis to identify survey effort, and the habitat walkover to identify potential electrofishing locations.

Power analysis informs on the confidence that a change, impact or effect can be observed, if they exist. It uses four parameters to calculate the statistical Power of the proposed test for a given sample design, thereby allowing the designer to assess whether more (or less) effort is needed, or to assist with understanding survey limitations. These parameters are:

- Effect Size – how big a change do we wish to detect;
- Sample Size – how many samples do we need to detect change at a given effect size;
- Variance – how variable is the subject we are testing (greater variance can often result in less confident results); and
- Significance level – usually 0.05 – the probability.

Figure 1.1. presents the three main channel restored areas in the programme. With the addition of a control group it was proposed that these four groups would be compared using an ANOVA type analysis to test differences between groups. Power Analysis was carried out in R, using the package "PWR" for several effect sizes (f = small, medium and large; Cohen, 1988) and k (number of groups) = 4, p = 0.05 and power (β) = 0.8 using historic data.

Effect size was calculated using a comparison between the trout densities from the Longcote and the Eddleston, to quantify potential variation between two distinct sites within the Eddleston catchment.

These differences were found to be large and assigned an effect size value of *f* = 1.04. When the Power Analysis was run using this value, *n* per group was 3.7. Given the budget (and subsequently issues regarding suitable areas for survey, - see below), a maximum of 3 samples per group could be facilitated, potentially allowing large differences to be identified.

Habitat walkover

Habitat walkovers were carried out in July 2017. This is outside the optimum window for habitat walkover (December to April) and extensive vegetation growth obscured some features.

The river was walked from Cringletie (NT 23814 44199) to Shiphorns (NT 24294 49781), over 6km, with a short section downstream near Peebles (which was identified as a location of sampling for invertebrates (NT 23916 43831 to NT 236020 43895). The walkover was carried out by Tommy McDermott, with attendance by James Hunt.

The field approach used was a combination of the methodology presented in the EA manual for salmonid habitat restoration (Hendry & Cragg-Hine, 1997), the mesohabitat conceptual approach utilised by the River Habitat Survey (RHS, Raven et al., 1997) and the information recorded as part of the SFCC electrofishing data collection. In its original format, the 1997 EA method provides a useful template for data recording, but restoration science has progressed since its production and this updated method reflects that progression.

Habitats and other relevant features were drawn onto expanded Ordnance Survey (OS GB carto © Crown Copyright and Database rights 2019) maps to create a quantified and pre-georeferenced spatial record. In some areas OS assets were not current and did not include the new channel. Best estimates using map features were used where this occurred (e.g. below Cringletie bridge). A complete inventory of possible features is presented below in Table 2.2, while additional data recorded within certain feature fields are shown in Tables 2.3 (bar forms) and 2.4 (modifications). Not all these features were found on the Eddleston during the survey.

Table 2.2. A complete inventory of potential features recorded by the walkover method used. For ease of understanding they are divided up into Feature Classes, with the Feature recorded linked to the Field Code used. The type of data (e.g. area – polygon, point and line) is also identified, indicating the output formats for GIS.

Feature Class	Feature	Field Code	Type
Flow biotopes	Free Fall	Ff	Area
	Chaotic	Ch	Area
	Cascade	Ca	Area
	Backwater	Ba	Area
	Broken water	Bw	Area
	Rapid	Ra	Area
	Chute	Ch	Area
	Upwelling	Up	Area
	Riffle (0+)	RIF	Area
	Run (<0+)	Ru	Area
	Glide (<0+)	Gl	Area
	Pool (adult, refuge)	Po	Area
	Impoundment	Imp	Area
	No perceptible flow	NPF	Area

	Exposed in-channel bedrock	BED	Area
	Dry channel	DRY	Area
In-channel features	Side Channel	SC	Area
	Side bar *	Sb	Area
	Mid channel bar*	MCB	Area
	Point Bar*	Pb	Area
	Island	Mi	Area
	Debris dams	Dd	Area
	Large woody debris	LWD	Point
	Ford	Fo	Area
	Culvert	Cu	Area
	Erratic boulders	Bo	Point
	Lithophilic spawning habitat	X	Point
	Lamprey juvenile habitat	L	Area
Modifications	Weir	W	Point
	Ford	Fo	Area
	Culvert	Cu	Area
	Channel reinforcement	Ri	Area
	Hard bank mods	HMB	Line
	Soft bank mods**	SMB	Line
Riparian	Overhanging trees	OH	Line
	Tunnel vegetation	TV	Line
	Bank erosion	Er	Line
	Poaching	Pc	Line
	Overhanging bank	OB	Line
Pollution inputs	Diffuse pollution source	DP	Area
	Diffuse pollution input	DP	Point/Line

***Table 2.3 For each bar form recorded, the dominant and subdominant substrate is recorded as linked observations to the main Feature. The Observation Code and definition are presented here.**

Observation Code	Composition
Bo	Boulder >256mm
Co	Cobble 64-256mm
GP	Gravel/pebble <64mm
Sa/Si	Sand/silt
Veg	Vegetated

Table 2.4 For each channel modification recorded, linked observations to the main Feature are recorded. The Observation Code and definition of these observations are presented here.

Feature Code recorded	Observation Code	Composition
Hard bank modification (Reinforcement)	Br	Brick (Br)
	Cc	Concrete (Cc)
	St	Stone (St)
	RR	Riprap
	Ga	Gabion
	TD	Tipped Debris
Soft Bank Modifications	Ra	Realigned
	Rs	Resectioned
	Emb	Embankment
	OW	Overwidened

Data digitisation and handling

Following completion of the walkover, the information was digitized using ArcGIS 10.2. The completed maps were reviewed, and several potential survey sites were proposed.

Site locations

The relevant information was combined on ArcGIS and several potential electrofishing sites proposed for control and treatment areas. This site locations were then verified pre-survey to ensure they met the requirements (vis-à-vis depth, flow velocity and width) for electrofishing. The sites are presented below in Table 2.5 and illustrated in Figure 2.3 overleaf.

Table 2.5. Electrofishing locations

Site	Location	X	Y
EDCon-1	Cringletie	324237	642747
EDCon-2	Lakewood	323793	645604
EDCon-3	Shiphorns	324267	648995
EDTR-01	Cringletie	323746	644446
EDTR-02	Cringletie	323762	644807
EDTR-03	Cringletie	323743	644953
EDTR-04	Lakewood	323796	645156
EDTR-05	Lakewood	323796	645177
EDTR-06	Lakewood	323776	645510
EDTR-07	Shiphorns	324281	649164
EDTR-08	Shiphorns	324289	649376
EDTR-09	Shiphorns	324300	649498

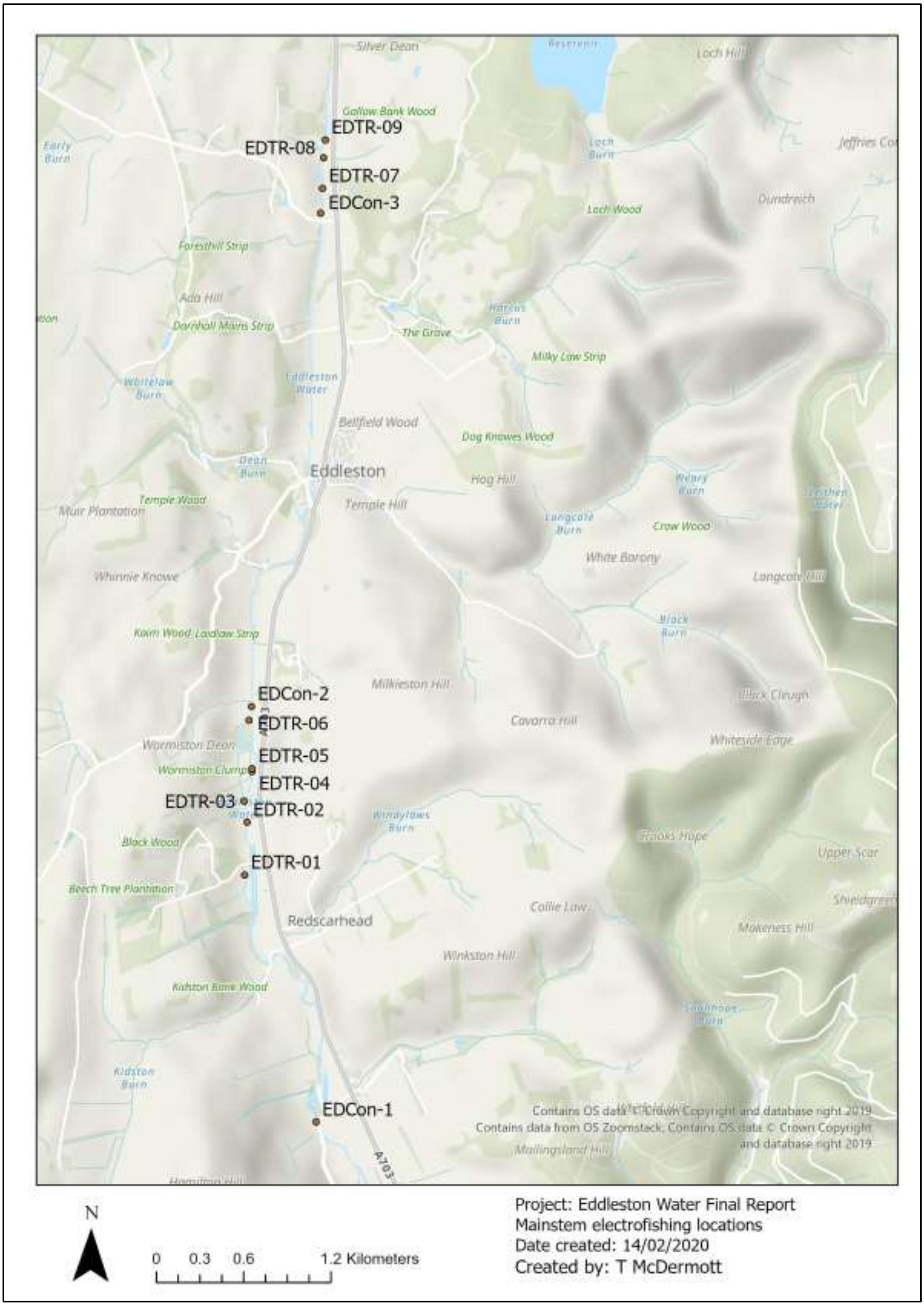


Figure 2.3. Mainstem electrofishing locations

Control sites were widely distributed within the catchment. The most downstream site, EDCon-1 was placed well outside the restoration area to allow a systematic control to be present and assist with alerting the project of any acute lack of fish entry into the Eddleston. This control site also coincides with a sediment monitoring trap just upstream, and the presence of the invertebrate monitoring point. Control sites were also placed within the Lakewood and Shiphorns areas (EDCon-2 and -3).

Treatment locations were placed within the restored areas in Cringletie, Lakewood and Shiphorns. In Lakewood particularly, site location was significantly constrained by an abundance of very deep pool and glide sections, which are unsuitable for the standard electrofishing approaches used here. Site locations were also limited by the total amount of restored channel within each area.

2.2. Electrofishing

Quantitative sampling of main channel sites in the Eddleston Water was carried out by a three-person team (led by the Tweed Foundation) following Scottish Fisheries Co-ordination Centre (SFCC) protocols[3] for fully quantitative fishing. This method was created specifically for capturing salmonids, but other fish species are caught. All in-channel surveys were led by the same operator (Kenny Galt), and habitat data was collected through the study by the same individual (James Hunt). This assisted with minimising sampler effects.

Site selection was guided by the habitat walkover (as per Section 2.1). Suitable areas were identified as being within the zone of interest with wadeable habitat of sufficient area to guarantee a sample is taken from approximately 100m^2 and sites positioned as far apart as possible. Site selection did pose some challenges due to the sizes of the restored areas and restrictions on habitat areas suitable for electrofishing.

Once a site was selected, approximately 100m^2 of channel was stop-netted and fished in a triple-run fully quantitative fashion with a single anode, banner net and two hand nets. Fishing conditions were good, with flows within the optimum boundaries and water clarity high. *Ranunculus* was present at many sites, potentially interfering with fish capture; however, the method used allows for some escape as it is expected that subsequent runs will recover almost all fish, with subsequent modelled density estimates accounting for any remaining. Fish were processed in a standard fashion and the length of fish recorded. The first 50 fish of each salmonid life history stage or species were also weighed.

Habitat data were collected as per the SFCC recording form. All data were entered from the field forms and stored into a bespoke spreadsheet with summary information (site characteristics, calculated fish densities and length summaries) stored in a master spreadsheet. Microsoft Excel was used for data storage and simple data description.

2.3. Redd surveys

Redd surveys were carried out on the main stem of the Eddleston led by Forth Rivers Trust and aimed to capture the spawning effort by salmonids within restored and unrestored areas. The approach is based on the American Salmonid Field Protocols Handbook (Johnson et al., 2007), and was refined during winter redd surveys of 2016/2017 in a Scottish river (River Forth Fisheries Trust 2017). The survey team was led by Tommy McDermott to ensure consistency of recording, with James Hunt, Colin Adams and Mike Voermans present during certain surveys.

The river was walked noting the presence of salmonid redds. Several associated characteristics were recorded, and the full suite of data collected is shown below in Table 2.6.

[3] Accessible here: https://www.sfcc.co.uk/training/electrofishing.html

Table 2.6 Information captured during redd surveys

Variable	Form
Date	Date
Depth	cm
Fish on	Yes/no
Flow type	Rif/Ru/Gl
Number of redds	count
Redd complex	Yes/no
Redd dimensions	cm
Species	BT/ST/AS
Visibility	1-10
Comments	
River width	m
Redd location	Head/tail/edge/middle

Data were recorded digitally on an Apple iPad compatible app developed using ArcGIS 10 (2017) and ArcGIS Pro (2018 and 2019) and the ESRI Collector application, ensuring automatic georeferencing. Data was uploaded to Forth Rivers Trust ESRI servers once signal was obtained and information displayed on ArcGIS desktop.

2.4. Data Analysis

Data treatment

Statistical analyses were carried out in R (R Core Team, 2019). Packages used will be referenced within the relevant subsections. As a first step, normality was tested for all datasets analysed with a Shapiro-Wilks test. Non-normal data were transformed for normality before further statistical analysis.

Relationship between fish population data and restoration.

One defining characteristic of salmonid life history is the relationship between distinct habitat niches and life history stage (Armstrong et. al, 2003), coupled with the migratory potential to access these habitats. Fry are capable of dispersing to other habitats (e.g. overwintering habitats) and juvenile salmonids move out of the Tweed catchment tributaries after their first winter, either as smolts (salmon and sea trout[4]) which go to sea or parr (trout) which move to the main stem. There may also be significant mortality at the overwintering stage which is a significant bottleneck for salmonid productivity (Glover et al., 2018). The results presented in Section 3 demonstrate a much lower density of fish older than fry when compared with fry suggesting emigration by >0+ juveniles and, therefore, only the fry stage was included in analyses.

The data collected was used in an attempt to define some of the causes of variation in salmon fry density and specifically to test the effect of the treatment of the site (control or re-meandered). To do this, a mixed effects statistical model was constructed, using sampling site (SITE), year of data collection (YEAR) and site type (TYPE - control or re-meandered treatment) as potential predictors of change. This was implemented in R using the lme() function. Mixed modelling allowed Site to be added as a random factor as the same site was sampled in multiple years.

[4] 65% of smolts in a nearby Tweed tributary are 1+ (The Tweed Foundation, unpublished).

Similar subsequent statistical models were constructed to examine drivers of average fish length and the variation in fish length for species and fish size classes.

Interrelationships between habitat variables

Habitat data from the SFCC recording form resulted in sufficient variables (77) to potentially cause over-parameterization of any tests, therefore the parameters were manually parsed to remove those which were either redundant or had a variance of zero. This resulted in a smaller dataset of 33 variables. This number was further reduced using a mixed PCA approach (Ade4, Dray & Dufour, 2007), which permits both categorical and numerical data to be included within the same ordination.

Six PCA components were then extracted based on visual estimation of the scree plot (a bar chart which illustrates the declining importance of each successive extracted component) and which together accounted for 65% of variation in the river channel habitat dataset. Each of the six sets of PC scores representing riverine habitat types were then tested for any differences between control and re-meandered sites (site Type) using a linear model or non-parametric alternative.

Relationship between habitat and fish populations

To test for the potential effects of river habitat on fish density a linear (lm()) model was developed to investigate the relationship between the six habitat PC and fish density. This was then repeated for fish length and fish length variance.

Redd data analysis

Using the redd map, recorded redd presence and area of redds were categorised into control and treatment groups as per the electrofishing dataset. To control for unequal sample channel lengths (non-restored channel length surveyed was much greater than restored channel), data from these groups were standardised by category total river length to produce the number of redds per kilometre of channel, and redd area per kilometre. To test for differences between redd data from control and treatment areas, a linear model was constructed using year and area (control, Cringletie, Lakewood or Shiphorns) as factors with relativised redd data as a response.

3. Results

3.1. Fish density

Atlantic salmon

Mean Atlantic salmon densities over the survey period per site are shown below in Figure 3.1.

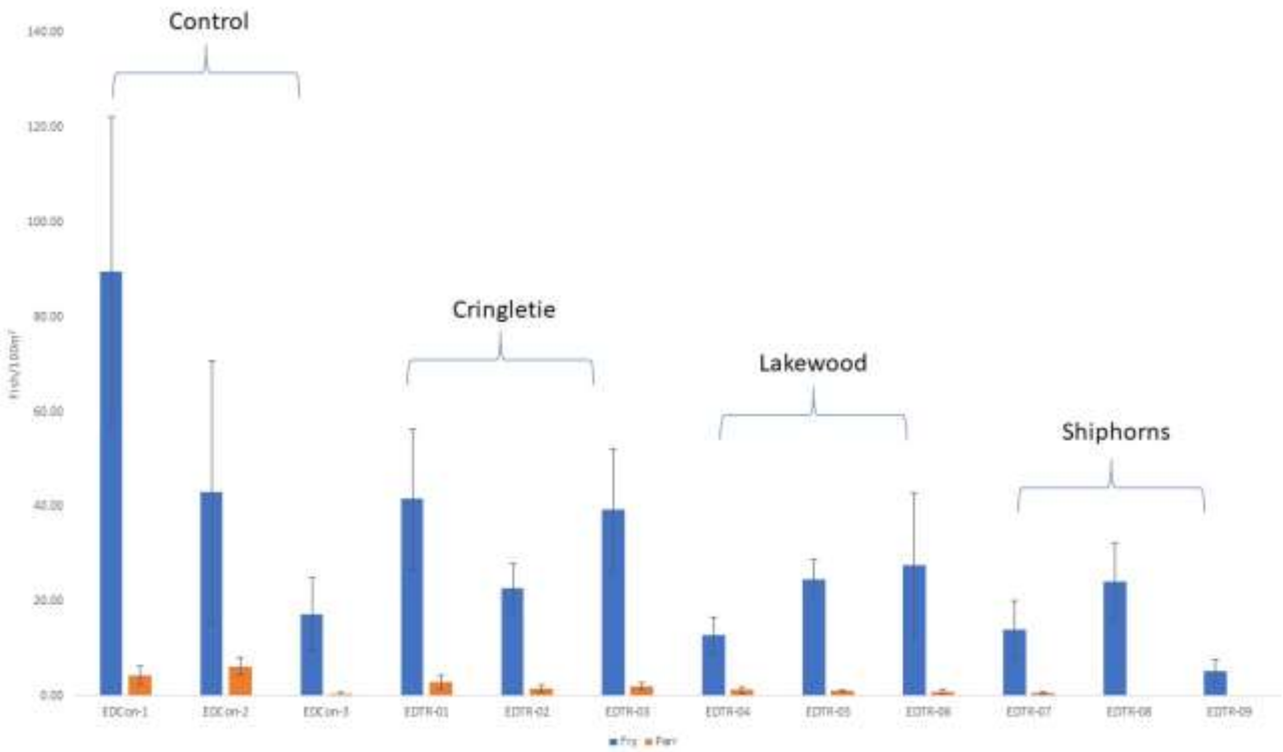


Figure 3.1. Average salmon (fry and parr) density (fish/100m^2) for each site, with SE bars, from 2017-2019, *n* per site = 3. General site locations are shown above, with site order the same for subsequent graphs. Cringletie Lakewood and Shiphorns sites are in order of downstream to upstream

River position did not appear to have a major bearing on fish numbers, and fry being most abundant at The EDcon-1 site (downstream/Cringletie control). However here, as at many sites, variance is large indicating between year differences in survey count. The decline in the number of older juvenile salmon is clear.

Salmon parr were not common, and densities greater than 10 parr/100m^2 were not encountered during the study. Zero catch for salmon parr was encountered at individual Shiphorns sites (Figure 3.1, EDTR-08 and -09). Due to the productivity of the Tweed catchment, salmon often smolt after their first winter (Tweed Foundation 2019) and therefore the low densities of salmon parr may simply represent a life history strategy. Salmon parr were subsequently removed from further analysis.

Brown trout

Mean brown trout densities over the survey period per site are shown below in Figure 3.2.

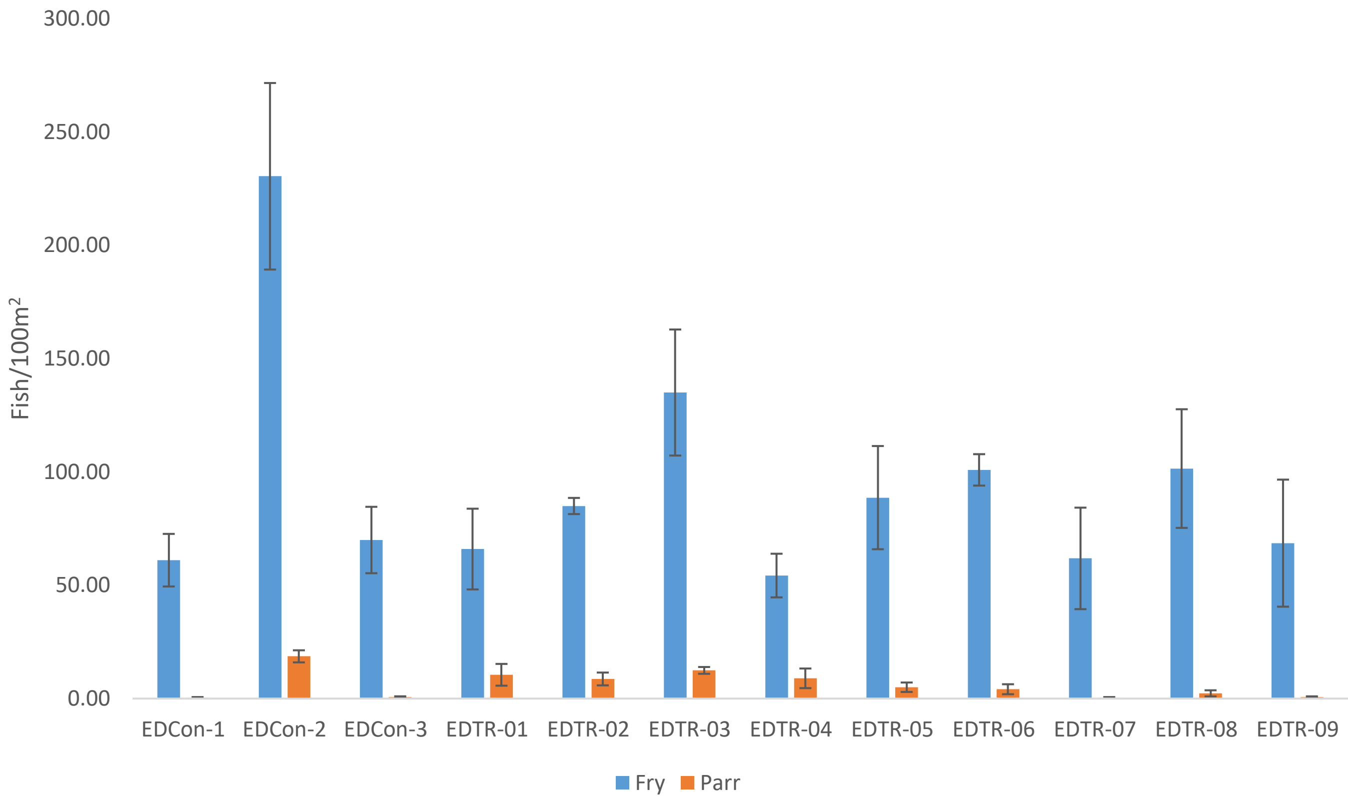


Figure 3.2. Average trout (fry and parr) for each site, with SE, from 2017-2019. *n* per site = 3. Note differing horizontal axis scale to Figure 3.1.

In common with the observed patterns for Atlantic salmon, density of fry was much higher than density of fish older than fry. The number of fish does not appear to be related to river position with Shiphorns (the most upstream survey areas) fry density potentially similar to densities at lower sites. Densities of older fish were higher at the mid-section of the study (EDcon-2, EDTR-01 to EDTR-06).

Atlantic salmon fry density – mixed model

To assess whether there were differences in the density of salmon fry between control and treatment locations a mixed modelling approach was used. Site type (TYPE – control or treatment) and YEAR were set as fixed factors, while the random effects due to site placement were controlled for by including SITE as a random factor. Salmon fry density data was square root transformed (Quinn & Keough, 2000) to allow the assumptions of normality to be met.

Mixed effect modelling (random = SITE, fixed = fry x TYPE + YEAR) of the transformed salmon fry data (square root) showed that there was a very strong and highly statistically significant effect of year on salmon fry density (2018, $df = 22$, $t = -4.05$, $p = 0.005$), but that whether fish density was from a control or re-meandered site did not significantly affect salmon fry density. This is illustrated below in Figure 3.3, showing the lower density in 2018.

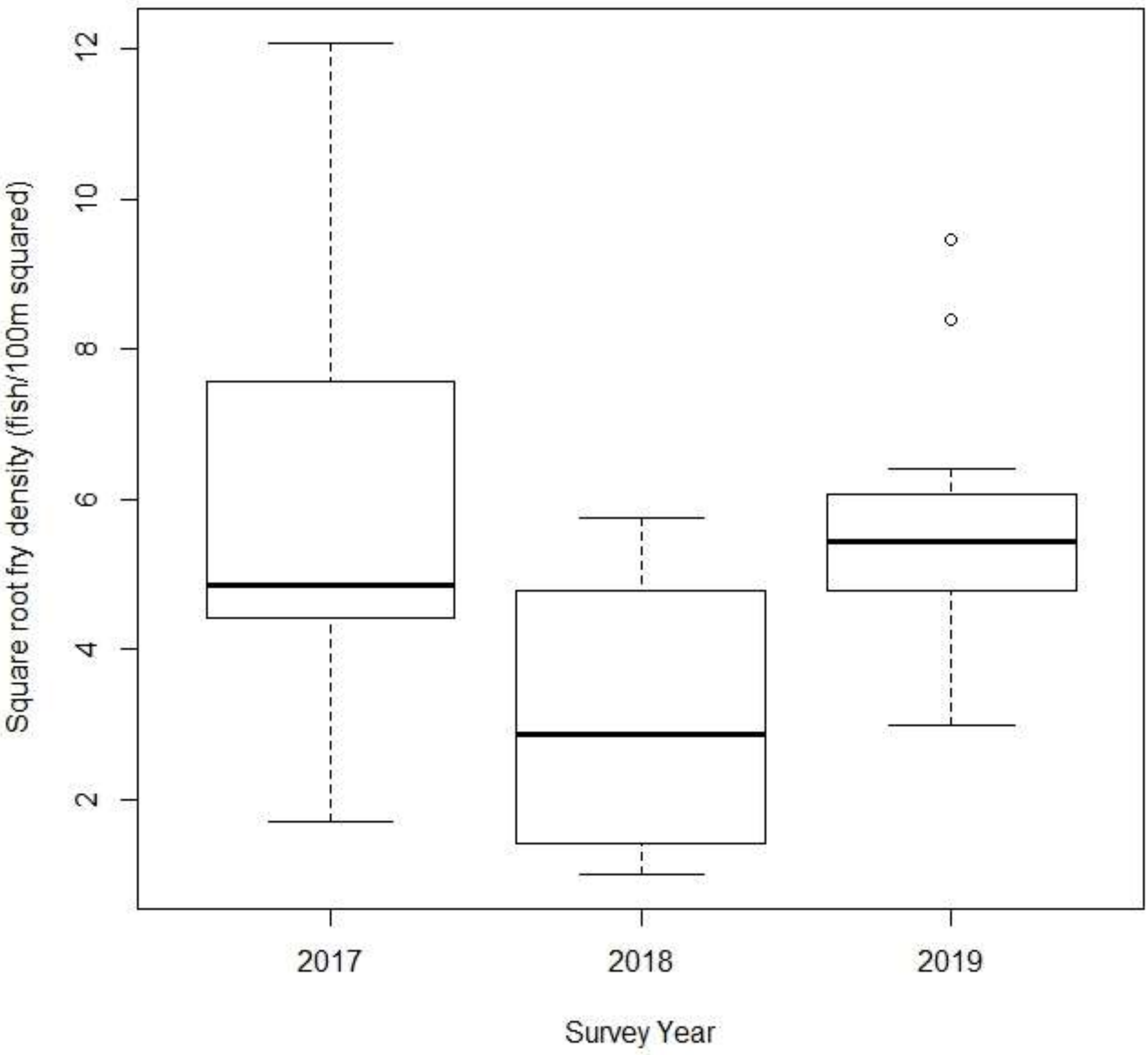


Figure 3.3. Square root transformed salmon fry density between study years. Whiskers represent maxima and minima, while the box limits represent the 25th and 75th %iles. The median is presented as the line within the box and outliers are presented as open circles.

This shows that once any random influence of SITE was removed, the biggest influence on salmon fry density was year, with density high in 2017 falling in 2018 and rising again in 2019. This suggests that inter-annual (environmental) factors are exerting a significant influence of salmon density.

Brown trout fry density – mixed model

Brown trout fry density was log10 transformed to meet normality assumptions.

Mixed effect modelling (random = SITE, fixed = fry x TYPE + YEAR) demonstrated no significant effect of Year, site Type (control or Treatment) or their interaction and log 10 density of trout fry. This shows that once the effect of repeated sampling at the same site were removed, the effect of YEAR and whether a sample came from a control or treatment location (TYPE) had no influence on trout fry numbers. This suggests that trout fry numbers were relatively stable across all sites and throughout the study period.

3.2. Fish Length

Atlantic salmon length

The mean lengths for salmon fry and older juveniles are shown below in Figure 3.4 with standard error bars.

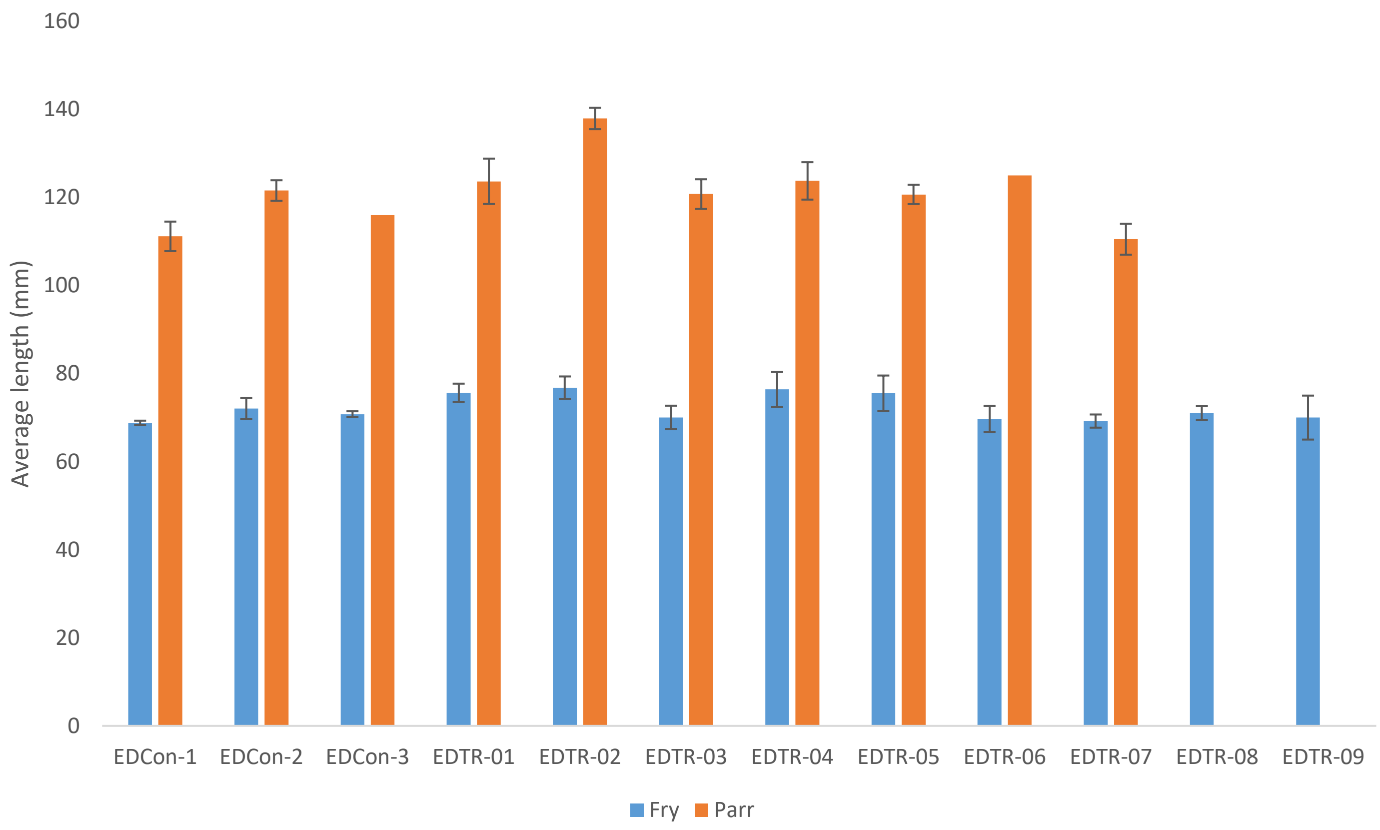


Figure 3.4. Average length of fry and salmon parr from control and treatment sites on the Eddleston 2017 – 2019. Bars represent standard error

Variance in average length between years did not appear large for either salmon fry or parr (where parr were caught). Due to the low sample size, values for parr may be influenced by a few very large or very small individuals.

Brown trout length

The recorded average brown trout lengths for the study period are shown overleaf in Figure 3.5. As with Atlantic salmon, numbers of fish older than fry were low, and therefore the values for average length may have been influenced by outliers. This is particularly relevant for brown trout where adult or subadult fish were captured, especially in 2019. An example fish is shown in Figure 3.6.

Fry average length across the study period does not appear to vary enormously between sites, based on site position or site type. There does seem to be a slight "hump" in the chart related to mid-section restored sites (Cringletie and Lakewood), potentially indicating larger fish in the >0+ group in these areas.

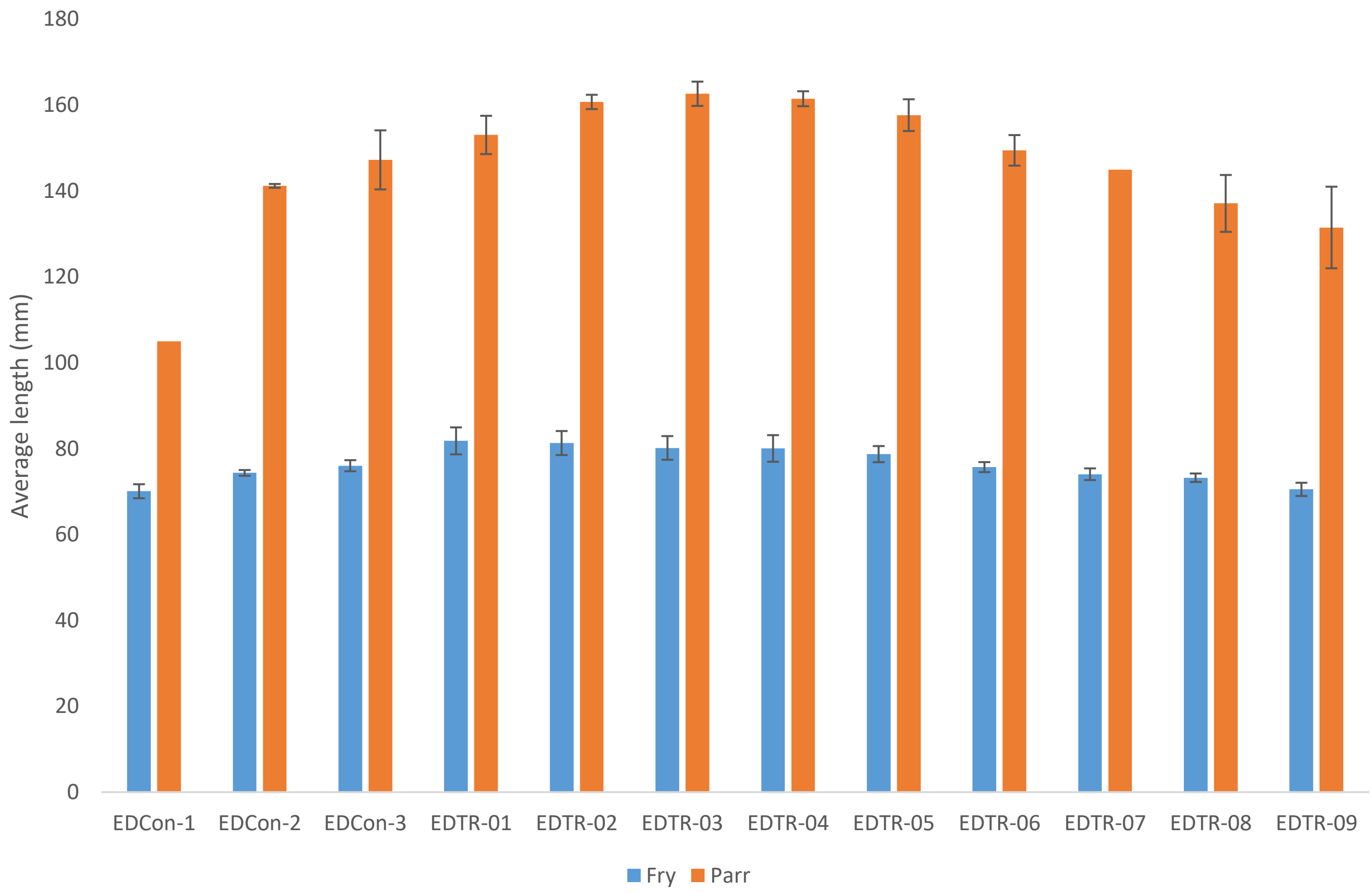


Figure 3.5. Average length of fry and trout parr from control and treatment sites on the Eddleston 2017 – 2019. Bars represent standard error. Note differing horizontal axis scale when compared with Figure 3.4.

Figure 3.6. Example large trout, caught in 2019

Atlantic salmon fry length – mixed model

Mixed modelling was used to determine any potential differences in fish length and variance in length (as measured by Coefficient of Variance, Quinn & Keough 2000) between control and treatment locations. No transformations were required.

SITE was once again used as a random effects term to control for multiple sampling at the same location, with YEAR and site Type retained as fixed factors.

Once the random effect of SITE was removed, there was no difference in salmon fry length between years (2017 – 2019) nor between site Type (control or treatment). There was a significant difference in salmon body length variance between years (2018, df = 22, t = -2.11, p = 0.047), with body length variance lower in 2018 than the other sampling years. This may reflect the reduced catch in salmon fry from that year, or potentially represent a reduction in intraspecific competition during a smaller population year.

Brown trout fry length – mixed model

SITE, YEAR and site TYPE were treated in a similar fashion to other tests presented above. Brown trout were highly significantly smaller in 2018 (df = 22, t = -2.95, p = 0.008), but there was no difference in trout size between control and re-meandered stream areas.

3.3. Habitat and fish metrics

Extracted components

Using a visual estimation approach 6 principal components were extracted accounting for 64.76% of the total variance in the habitat dataset. The sorted extracted component coefficients, with attributable variance for each component are presented below in Table 3.1. This permits the association between variables to be identified with a relative score.

Table 3.1. Extracted component coefficients from mixed PCA

Cumulative variance	**21.70%**	**33.28%**	**42.94%**	**51.33%**	**58.69%**	**64.76%**
Parameter	CS1	CS2	CS3	CS4	CS5	CS6
top complex RB	0.4	0.24	-0.1	0.01	-0.31	-0.12
bank face veg complex LB	0.37	0	-0.12	0.19	0.05	0.27
top complex LB	0.33	0.03	-0.14	0.15	0.09	0.14
cobble	0.32	0.03	-0.1	0.03	0.02	-0.06
boulder	0.27	0.02	-0.13	0.02	0.2	-0.14
Run flow	0.23	0.22	0.04	-0.17	-0.13	0.12
Riffle flow	0.23	0.1	-0.06	-0.06	0.02	-0.05
top simple RB	-0.13	-0.08	0.03	0	0.1	0.04
bank face veg simple LB	-0.15	0.11	-0.01	0.02	0.03	-0.09
top simple LB	-0.24	0	0.03	-0.13	0.1	-0.02
instream vegetation	-0.25	0.13	0.15	0.12	0.08	0.17
Macrophyte LB	-0.21	0.27	-0.15	0.03	-0.02	-0.06
Bare RB	0.19	-0.2	0.07	0.02	-0.12	0.13
gravel	-0.19	-0.22	-0.22	0.05	-0.1	-0.11
pebble	-0.09	-0.26	-0.18	-0.15	-0.24	-0.12

bank face veg uniform LB	-0.09	-1.15	0.75	-0.5	-0.52	-0.85
top uniform LB	0	-1.57	1.39	0.28	-0.8	-0.82
Improved grass	-0.23	0.37	0.45	0.42	-0.26	-0.38
Rock LB	-0.02	-0.22	0.36	0.19	-0.06	-0.07
Roots LB	0.14	-0.19	0.29	0.16	-0.1	0.03
other cons	-0.14	0.22	0.28	0.12	0.18	0.12
Area	-0.1	0.13	-0.23	0.06	-0.15	-0.02
smooth flow	-0.14	-0.22	-0.27	-0.06	-0.01	0.04
shallow pool	-0.13	-0.19	-0.27	0.18	-0.04	0.06
sand	-0.15	-0.14	-0.28	0.15	-0.03	-0.01
shallow glide	-0.04	-0.16	0.16	0.36	0.07	-0.12
Width	-0.04	-0.18	-0.15	0.29	0.11	0.18
Bare LB	0.17	-0.16	-0.05	-0.2	0.03	0.19
deep glide	-0.17	0.12	0.17	-0.21	0.12	-0.03
silt	-0.1	-0.02	-0.02	-0.27	-0.04	0.17
deep pool	-0.18	-0.02	-0.01	-0.28	0.06	-0.06
hall herb	-0.21	-0.12	0.08	-0.43	-0.14	0.03
bank face veg bare LB	-0.12	-0.21	-0.01	-1.33	-0.21	1.04
bank face veg simple RB	-0.05	-0.04	-0.01	-0.01	0.12	-0.01
scrub	0.15	0.03	-0.19	0.23	0.41	-0.32
Top bare LB	0.04	1.09	0.03	0.19	-2.35	-0.84
bank face veg complex RB	0.42	0.31	0.07	0.05	-0.99	0.07
undercut LB	-0.02	0.16	-0.08	0.21	-0.35	-0.15
Distance from Source	-0.17	-0.03	0.12	-0.04	-0.28	0.25
undercut RB	-0.1	0.18	-0.06	0.11	-0.24	-0.2
Broadleaf	0.26	-0.1	-0.15	0.07	-0.19	0.63
Macrophyte LB	-0.17	0.19	-0.06	0.17	0.07	0.21
Draped RB	0.02	0.03	-0.09	0.05	0.21	-0.28
Draped LB	0.01	-0.01	0.03	-0.24	0.1	-0.45

Component 1 (CS1 – 21.7% explained variation) is positively associated with indicators of higher stream power (riffle and run flows, cobble and boulder substrates), and negatively associated with an indicator of lower stream power (macrophytes). There is also a gradient of bank vegetation structure with indicators of complex vegetation structure positively related to CS1, while simple bank vegetation structure is negatively correlated with it.

Component 2 (CS2 – 11.6% explained variation) has a positive relationship with macrophytes, and a negative relationship with bare bank (potentially a proxy for erosion) and gravel and pebble substrates. This suggests another gradient with stream energy as an explanatory driver. There is also a negative association between uniform bank vegetation structure and CS2.

Component 3 (CS3 – accounting for 9.66% of explained variation) was positively associated with rocky and rooty banks, "other" substrates and improved grassland land use. It was negatively associated with shallow pool and glide (smooth) flows, site area and sandy substrate. This suggests that stream power is still an explanatory driver, with exposed banks on the positive end of the component gradient and slack flows and finer substrates on the other.

Component 4 (CS4 – 8.39% variation explained) is less clear than the previous components. There is a positive association with slacker flows (shallow glide) and channel width and a negative association with deep glide, silt and deep pool. There also appears to be some relationship between CS4 and bare banks (erosion, +ve) and tall herb land use and underdeveloped bank vegetation (bare and simple, -ve). Given the habitat features identified here, CS4 may be related to the Lakewood area.

Component 5 (CS5 – 7.36% variation explained) has a positive association with scrub land use and simple bank face vegetation, and a negative association with bare bank top, complex bank vegetation structure, undercut banks and distance from source. This component may be associated with the Shiphorns sites.

Component 6 (CS6 – 6.07% variation explained) demonstrates a positive relationship with broadleaf land cover and macrophytes, and a negative association with bank draped by vegetation (herbaceous). This suggests a cover/shading driver of this component.

Extracted component analysis

Extracted principle components scores (RS) were then compared between control and treatment sites to identify any quantitative differences in river habitat structures between restored and unrestored areas.

Only the scores related to the first component showed any difference between control and treatment (Wilcox sum rank test, $w = 176$, $p = 0.047$). Control sites had a stronger association with the first component than treatment sites. CS1 relates to a gradient based on bank vegetation naturalness (complex bank top and bank face vegetation – simple bank top and bank face vegetation) and in-channel energy (large substrates and riffle run flows – instream vegetation).

To permit a comparison between habitat components and fish density and size, a mean Component Score for each sampled electrofishing site was calculated. These mean scores were then modelled against the fish analysis parameters (density, length and length variance) using linear models. No relationship was found between the habitat components and fish density, length or length variance for either salmon or trout.

3.1. Redds

The distribution of recorded redds through the study period is shown overleaf in Figure 3.7.

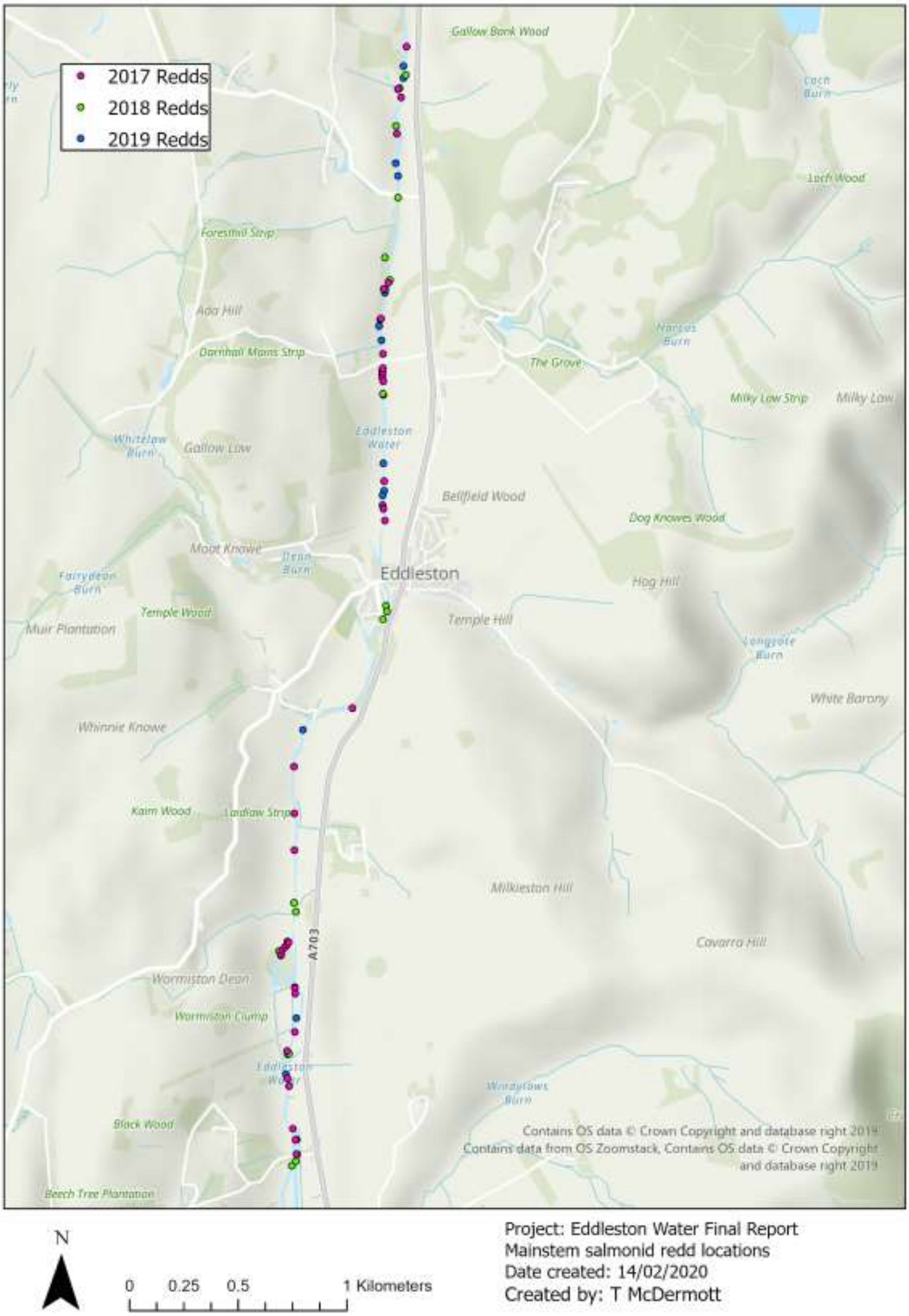


Figure 3.7. Distribution of recorded salmonid redds through the study period 2017 - 2019

The relative number of redds (redds per km of river) throughout the study period is presented below in Figure 3.8.

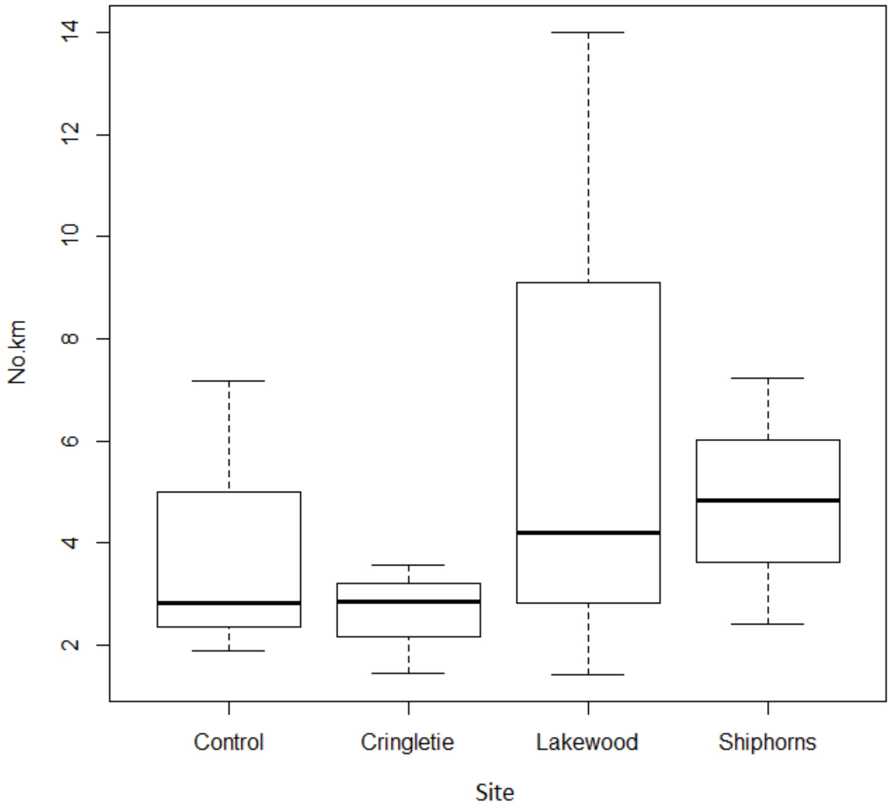


Figure 3.8. Relative count of redds observed during the study period 2017 -2018. Whiskers represent maxima and minima, while the box limits represent the 25th and 75th %iles. The median is presented as the line within the box.

To test for differences between redd counts from control and treatment areas, a linear model was constructed using Year and site Type as factors with relative redd number as a response. This model demonstrated there was no significant difference in relative redd counts between control and treatment locations. There was however a strong significant difference between years (2018, $t = -2.37$, $p = 0.047$; 2019, $t = -3.87$, $p = 0.005$) (Figure 3.9). Relative redd area was also modelled using the same factors (site Type and Year), with no significant effects of either variable or their interactions. This suggests that while the number of redds varies significant between years, this variation is driven by interannual effects and that redd numbers declined during the study (Figure 3.9).

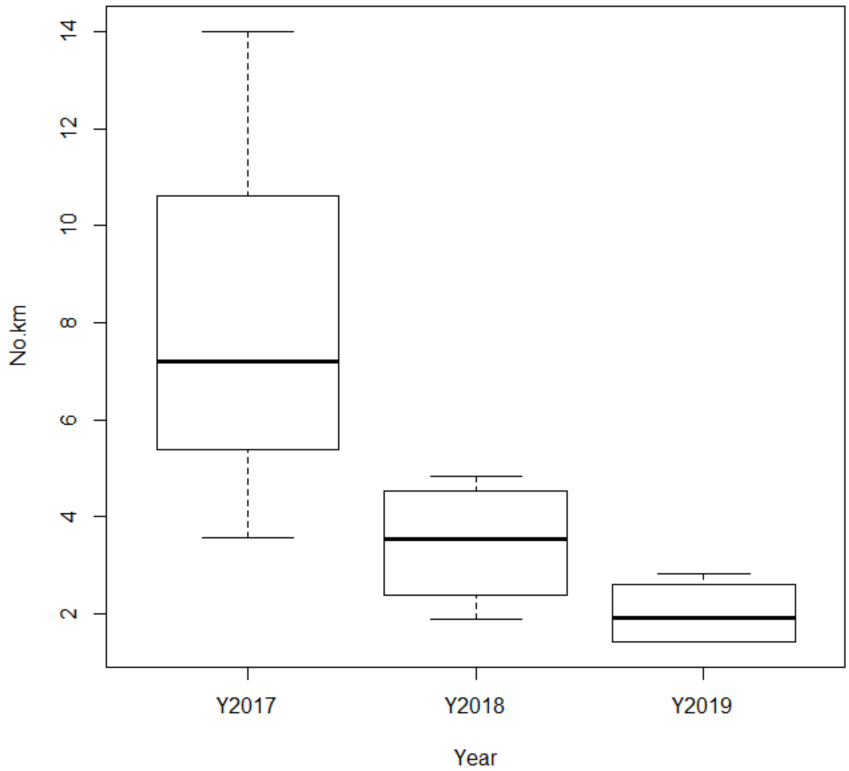


Figure 3.9. Relative number of redds by year during the study period (2017 – 2019)

4. Discussion

4.1. Caveats and limitations

Design and survey site location

BACI design is considered optimum for river restoration studies (Roni, 2005), however it was not possible to implement this framework here as the restoration works predate the beginning of fish monitoring by many years. Therefore, a Control-Treatment approach was used. This meant that control sites were subject to the same high-level interannual influences (e.g. rainfall – flow patterns and marine survival) on salmonid productivity as restored sites without a fixed pre-restoration benchmark dataset to assess relative responses to these high-level influences between control and impact sites, but this may have also been the case with a BACI design (depending on dataset size).

An additional factor which should be considered was the placement of the control sites. They were mostly located close to their linked restoration area and little change was observed over the three years by the electrofishing team. However, based on the findings of other monitoring presented by SEPA on channel sediments and channel geomorphology the influence of restoration measures may be extending beyond these restored areas. Geomorphological monitoring outputs showed change in channel form at unrestored areas indicating that control sections of the Eddleston may be on a parallel but lower intensity continuum of change (SEPA, Eddleston project meeting 2020, report in prep).

Survey

The method related to the annual electrofishing surveys is a robust, standardised and widely accepted approach for juvenile salmonid surveys. They were carried out during similar times of the year and with the same survey leaders. However, these methods are not optimal for other species (lamprey, eels and stickleback) and the outcomes of this report are necessarily focused on salmon and trout.

Rainfall during autumn and winter 2019 affected redd surveys, reducing the availability of surveying time, limiting water clarity during appropriately low flows and potentially disturbing and mobilising substrates causing redd infill before counting.

During mainstem habitat surveys it was observed by the survey team that the restored channel still appeared quite underdeveloped. Undercuts were rare due to underdeveloped woody vegetation on the bank top, and in-channel cover was only provided by macrophytes where present. Large woody debris (LWD) was not common. In general, the programme sampling period was drier than average (see Figure 4.1). This lack of hydrological force to drive channel change may have reduced the availability of new habitats for fish. A linked factor may be the absence of woody debris in most of the new channels (with the exception of parts of the Lakewood section). This absence is related to both the limited LWD use during construction and design, and the immaturity of riparian trees along most of the new channel. It may be that site recovery could be categorised into primary and secondary change, with the former represented by in-channel macrophyte growth and bankside tall herb growth and occurring reasonably rapidly, and the latter corresponding to longer-term change related to tree cover, LWD and undercut development.

Based on the *a priori* Power analysis, only large differences may have been observed and more subtle smaller effects may have been missed. The number of samples was limited by space (primarily) and also by budget and method. However, even with a larger budget (and using the methods deployed), it would have been difficult to maintain independence of sampling within Lakewood and Shiphorns if further sampling sites were planned (see Richer et al., 2019 with respect to limits of available space for fish monitoring as part of river restoration projects).

4.2. Salmonid populations and habitats

Fry density

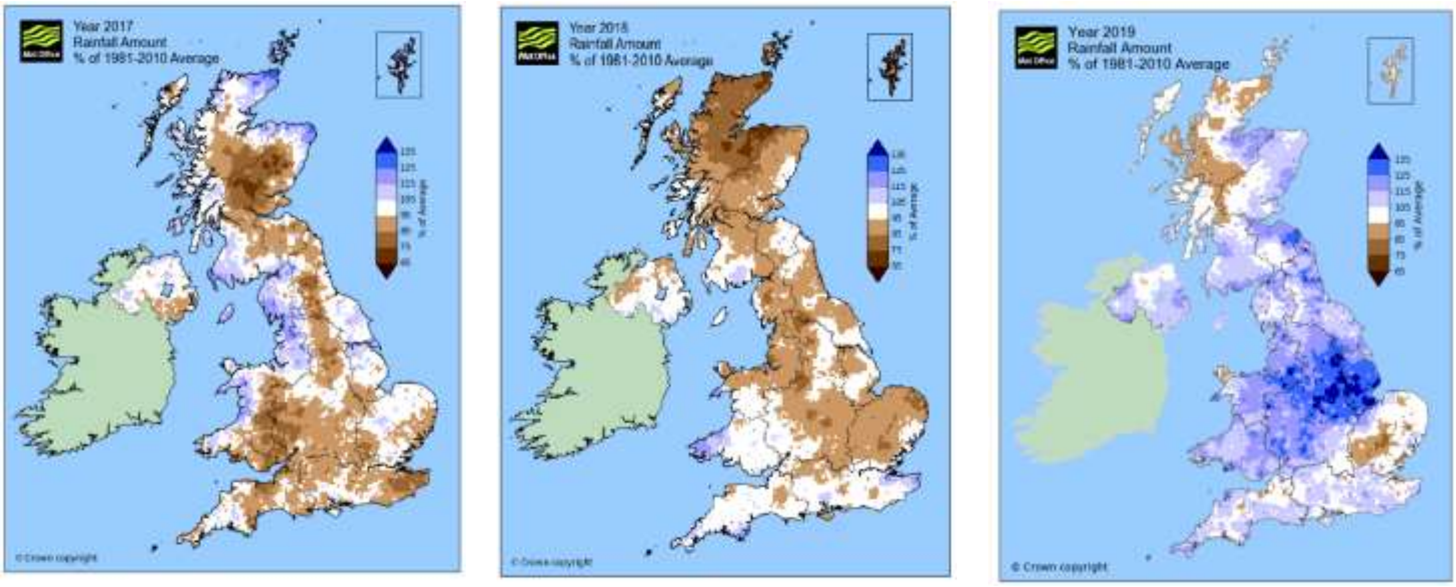
The main result of this study is that remeandering has not caused a reduction in the density of Atlantic salmon and brown trout fry within the Eddleston, based on fry density per $100m^2$. However, the area of channel restored is greater than the area of channel it replaced (by at least 10%, calculated using Ordnance Survey Carto digital maps); therefore, even considering not all new habitat is available for spawning and fry production, it is possible that the abundance of fry in the restored channel (from 2017 – 2019) is greater than would be present if the channel had not been restored. This concept is important but would need further refinement by assessing the specific habitat types and area in the new channel.

Based on the observed difference between historic data and data collected for the monitoring programme, the salmon fry densities recorded during the survey were broadly lower than those recorded during surveys pre-restoration, potentially reflecting the decline in salmon returns to the Tweed over recent years[5]. The reasons for this decline are complex, including cyclical patterns of relative abundances between multi-sea winter salmon and one-sea winter grilse (the dominant life history trait seems to switch every 60 years or so, with data suggesting another switch is currently underway, Tweed Foundation, 2018) and a major estimated decline in the number of returning salmon to Scotland (ICES salmon working group data, link here).

Based on Scottish catches (FMS, 2019), it appears that this monitoring programme has coincided with a decadal low in adult Atlantic salmon returns. Despite this, juvenile production (as measured in smolt output) can remain quite stable (McDermott et al., 2016; Gurney et al., 2007) and therefore it is possible that factors unrelated to adult returns contributed to the drop in salmon fry numbers in 2018. The results presented suggest that interannual factors were the main driver of between-year variance in both density and size of salmonids (where differences were observed), not the effect of restoration. It is very possible that very cold weather patterns in early spring 2018 contributed to the lower density of salmon fry. However, much of the variance in salmon populations can remain unexplained (Honkanen et al., 2019) and interpretation of interannually variable density data in aquatic species is often a challenge, especially where sampling effort is unrealistic or constrained (Sophie et al., 2019).

For brown trout density, there was no difference between years or control and treatment sites, suggesting a certain stability within the system for that species. Whether this reflects a long-term pattern, or a snapshot is not clear, however the densities recorded during this monitoring programme do not reflect the highest densities recorded during the surveys reviewed as part of the historic data assessment. Information on trout is not as well developed for the Tweed as for salmon, but current work being undertaken by the Tweed Foundation may eventually lead to pertinent information on trout population dynamics being presented in future years, particularly the inter-relationship between resident brown trout (trout that remain in freshwater) and sea trout.

[5] It is worth noting that on the 2020 Scottish Government Conservation of Salmon Assessment (Scottish Government, 2020) has proposed a Category 1 status for the River Tweed, indicating a healthy population of Atlantic salmon.



a. 2017 **b. 2018** **c. 2019**

Figure 4.1 a – c. Average rainfall values maps from 2017-2019 showing drier than average (browner) years 2017 and 2018, with slightly wetter (blue) than average year in 2019. © Met office and Crown reproduced under the Open Government Licence

Lengths

Another important result from this study is that remeandering has not caused a reduction in growth in the restored sections of the Eddleston. Atlantic salmon length variance was reduced in 2018, while brown trout fry were smaller in 2018. This differing results for salmon and trout could be related to the successive very cool periods in spring 2018 (beginning with the "beast from the east") potentially delaying ova emergence times and alevins growth of the earlier hatching trout. This may not have impacted on salmon total length due to their later emergence, but it may be the cause of their reduced density in 2018.

Habitats

There were differences in habitat between control and treatment sites. Recorded habitat features related to stream power and bank vegetation (and correlated to the first PCA axis) diverged between control and treatment sites. Control sites had a stronger association with a gradient described by larger substrate sizes, more developed bank vegetation, and run and riffle flows; and less instream macrophytes and simple vegetation. The assumed greater stream-power at straightened (unrestored) control sites can explain the larger substrates and faster flows, and the less abundant instream vegetation. However, it may take some time before the critical processes required to restore habitat-mediated ecological functionality are fully operational, for example physical meso- and micro- habitats may require colonisation with vegetative organisms before suitably habitat structure can be optimized (Muotka & Laasonen, 2002).

The association (and lack thereof) between sites and bank vegetation complexity probably relates to the fact that vegetation was cleared during construction of the new channels and has yet to reach the level of the more established vegetation at the undisturbed control locations.

Redds

Anecdotally spawning activity is often one of the first observed responses to restoration involving channel remeandering (H. Moir, pers. comm.) but possibly simply because construction should be carried out during warmer and drier periods in summer, which are then followed by the spawning period. There is some evidence of consistent improvement in spawning activity linked to restoration (e.g., Harrison et al., 2019), and fish from redds in restored areas may demonstrate improved survival (Wheaton & Pasternak, 2004). The analysis carried out here showed that there was a downward decline in redd numbers over the period of this study, while there was no difference in redd area. This suggests that larger, but fewer fish may have spawned in the Eddleston in later years. This reduction in spawning may conform with the estimated decline in returning adults, but methodological caveats, particularly important for 2019 data, mentioned previously should be observed when considering this result.

4.3. Restoration and fish

The wider restoration context

The remeandering of sections of the Eddleston was framed within the concept of ecological restoration; however, it was also a significant disturbance with a brand-new channel formed and an old but functional (from a salmonid perspective) channel closed-off, dewatered and filled in. Restoration should be considered a disturbance which promotes an unbalancing of eco-geomorphological processes and is potentially a gamble (Paillax et al., 2009); it is remarkable that within such a relatively short period of time the restored sections have reached productive parity with equivalent un-restored sections (based on density of fry per 100m^2). The restored channels have recovered and evidence from geomorphological monitoring carried out alongside ecological monitoring suggests further improvement is possible (SEPA, Eddleston project meeting 2020, report in prep).

The results here conform to many other single site/channel studies attempting to link river restoration to improvements in salmonid populations, namely that none or only minor differences were observed (Lepori et al., 2005, Pretty et al., 2003), and catchment scale effects can still override restoration (Leps et al., 2016). This may have been particularly difficult to distinguish in the Eddleston because the fish community (as measured by densities of salmon and trout) was in a good state before restoration, and that changes in the Tweed population of salmon in particular may overwhelm minor restoration-driven changes within small sections of a small tributary. Meta-analytical approaches using multiple projects do show positive responses from fish (Whiteway et al., 2010; Krail et al., 2015; Hasse et al., 2013; Marttila et al., 2019); however other metrics of measurement are often available to other rivers with more diverse fish communities and/or different survey methods.

However, it does potentially show the beginning of a divergence between habitats; this may be a salient indicator of change commonly seen in other studies (e.g. Krail et al., 2015; Lepori et al., 2005). Despite the time which has passed since sections of the Eddleston were remeandered, only a slight observable difference in habitats is present, and some of those differences are still related to the construction of the new channel. It may illustrate the time which is needed for restoration interventions to evolve (Krail et al. 2015; Hasse et al.; 2013) and mature to the point where channel habitat structures (and structural quality) depart from unrestored sections, with the subsequent improvements in fish population measurements. However, time alone may not be an accurate predictor of restoration success for fish (Roni, 2019) and restoration may need to be linked to other, wider-scale catchment processes such as general land use patterns and agricultural land use in particular (Leps et al., 2016).

The Eddleston project aimed to restore natural processes to a section of channel as far as could be managed given local constraints. Unlike other reach-scale restoration programmes (e.g. Allt Lorgy project in the Spey catchment), the restoration was still limited by stakeholder wishes and infrastructure. Therefore, not all desired interventions were possible. Where a more intensive restoration method was possible (e.g. Lakewood), change in channel habitat structures was seen due to the presence of established large trees and also the availability of LWD to the restoration design, and these structures are linked to fish density improvement in restoration projects (e.g. Polivka & Claeson, 2020). In other sections such as Cringletie there is an absence of hard features in the channel and along the bank that could have facilitated the development of undercut banks, pools and the provision of cover. Indeed, it seems that the only cover available is from *Ranunculus*, which is more related to restored than unrestored sites, and is seasonally variable. Ultimately there may have been a more evident salmonid community response with an increased use of fish-specific cover in the design (Finstad et al., 2007; Gargan et al., 2002; Nislow et al., 1999; Jong *et al.*, 1999).

Field Methods

The implementation of a three-year programme was a funding requirement as opposed to the optimal design, but met the minimum criteria of two or more years sampling recommended for fish and restoration projects (O'Neal et al., 2016), but a short-term programme may be subject to the continual evolution of the channel post-restoration (Wohl et al., 2005). In addition, strong interannual variation in fish densities dominated the data, potentially masking other more subtle effects. Limitations on the available space for survey may have constrained the project outcomes (Richer et al., 2019) and survey design may also have been at the threshold for detecting change.

Issues relating to continual post-restoration channel evolution and limited space may have been exacerbated by the methods used. To ensure a standardised electrofishing approach consistent with other projects in Scotland, fully quantitative electrofishing was carried out. This approach is subject to several assumptions; namely the availability of approaching $100m^2$ site area; an independence of sampling locations; and wadeable habitats. The approach is also tailored to mobile salmonids, and the confidence in the method to establish quantitative population estimates for more cryptic fish species (eel, lamprey spp. and stone loach) is low. Therefore, the areas where pool formation was a feature of the post-restoration channel evolution could not be surveyed as they were too deep, and fish communities associated with those habitats could not be assessed.

This suggests that there exists a requirement to develop a restoration-specific fish monitoring tool. This would allow future projects to confidently assess the response of other species (eel, lampreys, etc) as well as older classes of salmonid (resident adult trout) to restoration, and also extend the sampling to other, deeper habitats which were not surveyed here. This may require a multiple approach using semi-quantitative electrofishing for salmonids (e.g. Kennedy & Crozier 1994), Point Abundance Sampling (Scholten, 2003; Harvey & Cowx, 2003) for eel and lamprey, or even angling (e.g. Pinder et al., 2019) for larger fish. A pre and post restoration monitoring habitat walkover is also something which could be included, specifically where project outcomes suggest a change in productivity based on absolute area of channel (as here). This could be coupled with invertebrate, macrophyte and physical process monitoring to provide a cohesive and consistent ecological monitoring approach for future restoration projects.

Redd surveys are one of the few ways to assess adult system penetration as well as (potentially) abundance in relation to salmonid reproductive behaviour and habitat use. In many rivers where land-use inputs are restricted by an absence of intensive agriculture, remoteness or the presence of wooded buffers, channels can remain remarkably clear even at relatively high flows; however, on the Eddleston rising flows were often accompanied by discoloration. This was common in 2019 and resulted in a possibly compromised data collection as redds simply could not be seen or were rapidly filled in by substrates mobilised by the high flows. Therefore, as a standalone monitoring approach redd surveys may not be robust enough. However, when coupled with other fish monitoring methods, redd surveys can provide semi-quantitative supporting information which could assist with understanding results from these other methods.

5. Conclusions and the future

The project has recorded a recovery from significant in-channel intervention by salmon and trout populations, with disturbed and undisturbed locations hosting similar densities of fry of similar sizes. It is also possible that, based on a potentially greater area of restored channel compared with the original channel, salmonid fry abundance may have increased. It has also shown a potential divergence of control v restored location habitats, which if on a continuum of change, may eventually impact on recorded fish densities. The channel may not have evolved suitably (due to an absence of hydrological force and hard in-channel structures) to the point where fish populations are influenced; however, the sampling programme may also require more time.

While recognising the presence of constraints, the limitations of the channel design with specific respect to improving fish habitats may have been exposed. The guiding process restoration principle (Beechie et al., 2010) can lead to long-term stability in restored rivers but may be unsuitable for rapid improvement in fish habitat. Process restoration can need time for new channels to evolve and for the drivers of natural river processes to influence the channel (e.g. trees must grow before they shed boughs which form LWD accumulations). An increased use of instream LWD structures would have bridged the gap in time between completion of works and the point where natural vegetation growth, working in combination with other river processes, could facilitate improvements fish habitats.

Therefore, it is important that the monitoring programme continues, and the method should be retained for this project along with an acceptance that the study will only be able to assess changes to salmonid fry communities. A repeat of the habitat walkover after significant periods of rainfall (e.g. winter 2019/20) may also assist in determining the rate of change in habitats and consequential change in salmonid fish populations.

Where future river restoration plans are proposed, it may be necessary to develop a specific and novel fish monitoring method which permits the capture of data for all relevant species and life history stages and their related habitat parameters.

6. References

Armstrong JD, Kemp PS, Kennedy GJA, Ladle M and Milner, NJ. 2003. Habitat requirements of Atlantic salmon and brown trout in rivers and streams. *Fisheries research*. **62**:143-170

Bacon PJ, Malcolm IA, Fryer RJ, Glover RS, Millar CP and Youngson AF. 2015. Can conservation stocking enhance juvenile emigrant production in wild Atlantic salmon (*Salmo salar)*. *Transactions of the American Fisheries Society*. **114**:642-654

Beechie TJ, Sear DA, Olden JA, Pess GR, Buffington JM, Moir H, Roni P & Pollock MM. 2010. Process-based Principles for Restoring River Ecosystems. *BioScience*, **60**:201-222

Cohen J. 1988. *Statistical power analysis for the behavioural sciences.* 2nd Ed. Lawrence Erlbaum, New Jersey.

Crozier WW & Kennedy GJA. 1994. Application of semi-quantitative electrofishing to juvenile salmonid stock surveys. *Journal of Fish Biology*. **45:**159–164.

Dray S & Dufour A. 2007. “The ade4 Package: Implementing the Duality Diagram for Ecologists.” *Journal of Statistical Software*. **22**:1–20

Einum E., Nislow KH, Reynolds JD & Sutherland WJ. 2008 Predicting population responses to restoration of breeding habitat in Atlantic salmon. *Journal of Applied Ecology*. **45**:930-938

Finstad AG, Einum S, Forseth T & Ugedal O. 2007. Shelter availability affects behaviour, size dependent and mean growth of juvenile Atlantic salmon. *Freshwater Biology*. **52**:1710-1708.

FMS. 2019. *Fisheries Management Scotland Annual Review 2019*. 40pp.

Gargan P, O’Grady M, Delanty K, Igoe F & Byrne C. 2002. *The effectiveness of habitat enhancement on salmon and trout stocks in streams in the Corrib Catchment.* In Proceedings of the 13th International Salmonid Habitat Enhancement Workshop. Edited by M. O’Grady. Central Fisheries Board, Dublin, Ireland.

Glover RS, Fryer RJ, Soulsby C, Bacon PJ & Malcolm IA. 2018. Incorporating estimates of capture probability and river network covariance in novel habitat – abundance models: Assessing the effects of conservation stocking on catchment-scale production of juvenile Atlantic salmon (Salmo salar) from a long-term electrofishing dataset. *Ecological Indicators*. **93**:302-315

Gurney WSC, Bacon PJ, Tyldesley G, & Youngson AF. (2008). Process-based modelling of decadal trends in growth, survival, and smolting of wild salmon (*Salmo salar*) parr in a Scottish upland stream. *Canadian Journal of Fisheries and Aquatic Science*. **65**:2606-2622.

Harrison JG. 2012. The Eddleston Water: Historical change in context. Report prepared by Historical Service for the Tweed Forum.

Harrison LR, Bray E, Overstreet B, Legleiter CJ, Brown RA, Merz JE, Bond RM, Nicol CL & Dunne T. 2019. Physical controls on salmon redd site selection in restored reaches of a regulated, gravel bed river. *Water Resources Research*. **55**:1-25.

Harvey J. & Cowx I. 2003. Monitoring the River. Brook, and Sea Lamprey, *Lampetra fluviatilis*, *L. planeri* and *Petromyzon marinus*. Conserving Natura 2000 Rivers Monitoring Series No. 5, English Nature, Peterborough

Hasse P, Hering D, Jahnig SC, Lorenz AW & Sundermann A. 2013. 2013. The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates and macrophytes. *Hydrobiologia*. **704**:475-488

Heggenes J & Gunnar Dock J. 2001. Contrasting temperatures, waterflows, and light: seasonal habitat selection by young Atlantic salmon and brown trout in a boreonemoral river. *Regulated Rivers: Research & Management*. **17**:623-635.

Hendry K. & Cragg-Hine D. 1997. *Restoration of Riverine Salmon Habitats; A Guidance Manual*. Fisheries Technical Manual 4, R & D: Technical Report W144, Environment Agency, Bristol.

Honkanen HM, Boylan P, Dodd JA & Adams CE. 2019. Life stage-specific, stochastic environmental effects overlay density dependence in an Atlantic salmon population. *Ecology of Freshwater Fish*. **28**:156-166.

Johnson DH, Shrier BM, O'Neal JS, Knutzen JA, Augerot X, O'Neill TA & Pearsons TN. 2007. *Salmonid field protocols Handbook: Techniques for assessing status and trends in salmon and trout populations.* American Fisheries Society and State of Salmon. Portland ON.

Jong MVZD, Cowx IG & Scruton DA. 1997. An evaluation of instream habitat restoration techniques on salmonid populations in a Newfoundland stream. *Regulated Rivers-Research and Management*. **13**:603-614

Kemp P. 2010. *In-channel placement of structures to enhance habitat complexity and connectivity for stream-dwelling salmonids*. In Salmonid Fishes: Freshwater habitat management. ed. P. Kemp. Blackwell

Krail J, Brabec K, Poppe M & Januschke K. 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. *Ecological Indicators*. **58**:311-321

Liermann M & Roni P. 2008. More sites or more years? Optimal study design for monitoring fish response to watershed restoration. *North American Journal of Fisheries Management*. **28**:935-943.

Lepori F, Palm D, Brannas E & Malmqvist B. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications*. **15**:2060-207.

Leps M, Sundermnann A, Tonkin JD, Lorenz AW & Haase P. 2016. Time is no healer: increasing restoration age does not lead to improved benthic invertebrate communities in restored river reaches. *Science of the Total Environment.* **557/558:**722-732

Marttila M, Louhi P, Huusko A, Vehanen T, Maki-Petays A, Erkinara J, Syrjanen JT & Muotka T. 2019. Synthesis of habitat restoration impacts on young-of-the year salmonids in boreal rivers. *Reviews in Fish Biology and Fisheries.* **29**:513-527

McDermott T, Baker A, Bostock J, Bull C, Clayburn C, Girvan J & Prodohl P. 2016. *River Dee Hatchery Appraisal*. Technical report prepared for the Dee Rivers Trust. 55pp.

River Forth Fisheries Trust. 2017. *The observed distribution of salmonid redds in channel of the Teith, Forth and Devon catchments, Forth District Salmon Fishery Board area 2016/2017.* Technical Report to the Forth DSFB

Miller DJ, Burnett KE & Benda LE. 2008. Factors controlling availability of spawning habitat for salmonids at the basin scale. *American Fisheries Society Symposium*. **65**:103-120.

Muotka T. & Laasonen P. 2002. Ecosystem recovery in restored headwater streams: the role of enhanced leaf retention. *Journal of Applied Ecology*. **39**:749-759

Nislow KH, Folt CL & Parrish DL. 1999. Favorable foraging locations for young Atlantic salmon: application to habitat and population restoration. *Ecological Applications*. **9**:1085-1099.

O'Neal JS, Roni P, Crawford B, Ritchie A & Shelly A. 2016. Comparing stream restoration project effectiveness using a programmatic evaluation of salmonid habitat and fish response. *North American Journal of Fisheries Management* **36**:681-703.

Paillax A, Doledec S, Castella E & Merigoux, S. 2009. Large river floodplain restoration: predicting species richness and trait responses to the restoration of hydrological connectivity. *Journal of Applied Ecology.* **46**:250-258.

Palmer MA, Menninger, HL, & Bernhardt E. 2009. River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice. *Freshwater Biology.* **55:**1–18.

Pinder AC, Harrison AJ & Britton R. 2019. Temperature effects on the physiological status and reflex impairment in European grayling *Thymallus* from catch-and release angling. *Fisheries Research.* **211**:169-175.

Pretty JL, Harrison SSC, Shepherd DJ, Smith, Hildrew AG, & Hey RD. 2003. River Rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology*. **40**:251-265

Polivka CM & Claeson SM. 2020. Beyond redistribution: Instream habitat restoration increases capacity for young of the year chinook salmon and steelhead in the Entiat river WA. North American *Journal of Fisheries Management*. In Press.

Quinn G, & Keough M, 2002. *Experimental Design and Data Analysis for Biologists.* Cambridge University Press.

R Core Team (2019*). R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria.

River and Fisheries Trusts of Scotland. 2014. *Data supporting Site Condition Monitoring of Atlantic salmon SACs*. SNH commissioned report 755. 352pp

Richer EE, Gates EA, Kondratieff MC & Herdrich AT. 2019. Modelling changes in trout habitat following stream restoration. *River Research Application*. **35:**680–691

Roni P. 2019. Does river restoration increase fish abundance and survival or concentrate fish? The effects of project Scale, Location and fish life history. *Fisheries*. **44:**1 7-19

Roni P, Pess G, & Morley S. 2010. *Monitoring salmon stream restoration: guidelines based on experience in the American Pacific northwest*. In Salmonid Fishes: freshwater habitat management. ed. P. Kemp. Blackwell

Roni P, Beechie TJ, Bilby RE, Leonetti FE, Pollock MM & Pess GR. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management*. **22**:1-20.

Roni P. 2005. *Monitoring Stream and Watershed Restoration*. Northwest Fisheries Science Center. 337pp

Scholten M. 2003. Efficiency of point abundance sampling by electro-fishing modified for short fishes. *Journal of Applied Ichthyology.* **19**:265-277.

Scottish Government. 2020. *Conservation of Salmon – Assessment for the 2020 season – Proposed gradings*. Consultation and Application of Conservation Limits 2020. Scottish Government.

Stewart-Oaten A, Murdoch WW & Parker KR. 1986. Environmental Impact Assessment: "Psuedoreplication" in Time? *Ecology*. **67**:929-940

Solazzi MF, Nickelson TE, Johnson SL & Rodgers JD. 2000. Effects of increasing winter rearing habitat on abundance of salmonids in two coastal Oregon streams. *Canadian Journal of Fisheries and Aquatic Sciences*. **57**:906-914

Sophie CF, Verena T, Martin D, Anthony M, Herve C, Jean-Michel O, Jeremy L, Barnard C & Nicolas L. 2020. Interpretations of interannual variability in long-term aquatic ecological surveys. *Canadian Journal of Fisheries and Aquatic Science*. In Press.

The Tweed Foundation. 2013a. *BASELINE ELECTRO-FISHING RESULTS 2014 for monitoring sites established on the Longcote Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

The Tweed Foundation. 2013b. *BASELINE ELECTRO-FISHING RESULTS 2014 for monitoring sites established on the Shiplaw Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

The Tweed Foundation. 2014a. *ELECTRO-FISHING RESULTS 2014 for monitoring sites established on the Longcote Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

The Tweed Foundation. 2014b. *ELECTRO-FISHING RESULTS 2014 for monitoring sites established on the Shiplaw Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

The Tweed Foundation. 2015a. *ELECTRO-FISHING RESULTS 2015 for monitoring sites established on the Longcote Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

The Tweed Foundation. 2015b. *ELECTRO-FISHING RESULTS 2015 for monitoring sites established on the Longcote Burn to monitor the effect of riparian fencing and tree planting on the local Trout population*. Report by The Tweed Foundation, Drygrange.

Werrity A, Ball T, Spray C, Bonell M, Rouillard J, Archer N, Bowles C & Moir H. 2010. *Restoration Strategy: Eddleston Water Scoping Study.* Report prepared for the Tweed Forum.

Wheaton JM & Pasternak GB. 2004. *Use of habitat heterogeneity in salmonid spawning habitat rehabilitation design*. Fifth international symposium on Ecohydraulics. Aquatic Habitat: Analyses and Restoration. Madrid, 2004.

Whiteaway SL, Biron PM, Zimmermann A, Venter O & Grant JWA. 2010. Do in-stream restoration structures enhance salmonid abundance? A meta-analysis. *Canadian Journal of Fisheries and Aquatic Sciences.* **67**:831-841

Wohl E, Angermeier PL, Bledsoe B, Kondolf GM, MacDonnell L, Merritt DM, & Tarboton D. 2005. River restoration. *Water Resources Research*. **41**:1–12.

Appendix A – Site images (2019)

EDCon-1

EDCon-2


EDCon-3


EDTR-01

EDTR-02

EDTR-03


EDTR-04


EDTR-05

EDTR-06

EDTR-07

EDTR-08

EDTR-09

Appendix B – Longcote Burn

B1. Introduction

In 2013, extensive sections of the Longcote Burn (a small second order Eddleston tributary) were planted with native trees, and a substantial buffered riparian was area isolated from surrounding farmland and related grazing pressures. To observe potential responses from the fish community, the Tweed Foundation began surveying this channel using a standard semi-quantified approach (The Tweed Foundation, 2013a, 2014b & 2015c) which was then used by Forth Rivers Trust when delivering the requirements of the Eddleston fisheries monitoring programme.

B2. Method

Survey

The sites surveyed are shown overleaf in Figure B1. There was a total of 12 sites surveyed from 2013-2015 and 2017-2019, with a control, unfenced sample, taken at a site from 2016-2019. Semi-quantitative sampling involves electro-fishing for a set length of time, in this case 3 minutes, while continuously working upstream. The advantage of this method is that with a short sampling period for each site, a high number of sites can be quickly established that provide a broad geographical coverage of the project area. Surveys were carried out in 2013, 2014 and 2015 by the Tweed Foundation, with surveys in 2017, 2018 and 2019 carried out by the project team (Forth Rivers Trust and Tweed Foundation).

The site was identified and fished in an upstream fashion for a total of three minutes active anode time. Fish were captured using handsets or using a banner net where the site was wide enough. Captured fish were placed in a bucket and processed using standard approaches (see main text), namely subdued with anaesthetic, speciated, measured and weighed, and returned to the site upon completion site survey.

Basic physical characteristics of the watercourse were recorded on the SFCC data recording form, including site lengths and widths, permitting the determination of site areas and thus a minimum density estimate of fish caught. Other features such as flow type and substrate, as well as riparian features were recorded.

Data description and analysis

Estimated fish density of each life history stage and species was calculated using measured area and catch, standardised to $100m^2$. However, total area fished was often a very small fraction of $100m^2$ (usually 10 – 20%) which led to a very large increase from actual count value to density value. This also potentially artificially amplified the sampler/sampling error and also the inherent variability in the system, so further analysis was carried out using abundance data only. Salmon were very rare, while the concerns relating to the use of fish older than fry described within the main text also existed on Eddleston tributaries. Therefore, only brown trout fry abundance was used as a response variable.

Data from 2013-2015 was provided by The Tweed Foundation and added to the overall dataset. A time series analysis on each site was carried out using the conservative Kendall-tau correlation test thus controlling for non-normality (Quinn & Keogh, 2002). Tests were carried out in R using Package "Kendall" (McLeod 2015) using the Kendall() function. To avoid spurious comparisons between the larger treatment dataset and the shorter control dataset, the latter was removed from further analysis.

Figure B1. Longcote electrofishing sites

B3. Results

Survey results

Brown trout parr and fry were commonly captured. Only three salmon were caught through the entire study period (two in 2017 and one in 2019) so that species was excluded from further comment. No other species were recorded.

The numbers of fish caught of the 2016-2019 surveys are shown below in Table B1, while the calculated densities are presented in Figure B2 and B3.

Table B1. Catch of brown trout fry and parr and older (“Older”) from the Longcote 2017-2019

Site	2017		2018		2019	
	Fry	Older	Fry	Older	Fry	Older
L1	12	2	5	2	4	1
L2	17	3	4	0	7	1
L3	12	2	11	2	3	2
L4	36	1	4	0	8	0
L5	43	7	3	2	7	0
L6	31	1	11	0	10	0
L7	41	10	15	1	13	0
L8	26	2	13	0	10	0
L Control	37	1	7	2	10	0
L9	26	4	10	3	13	0
L10	36	6	18	4	16	0
L11	26	6	12	6	5	1
L12	24	0	9	0	8	2

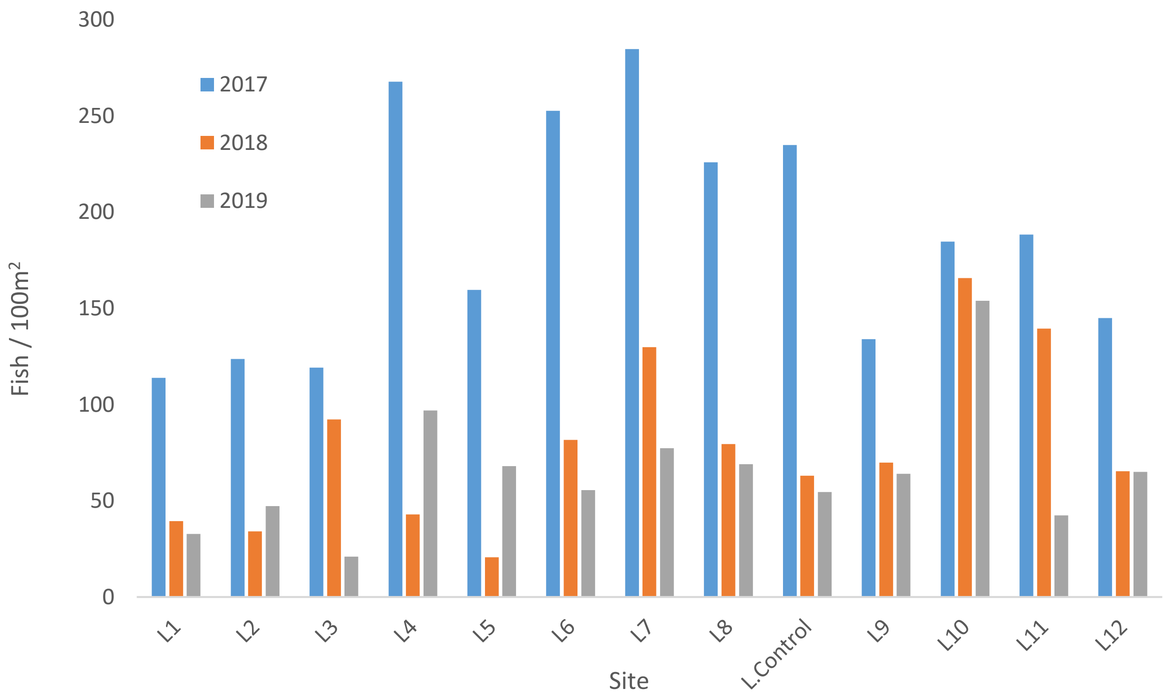


Figure B2. Calculated densities (minimum density estimate) of brown trout fry from Longcote 2017-2019

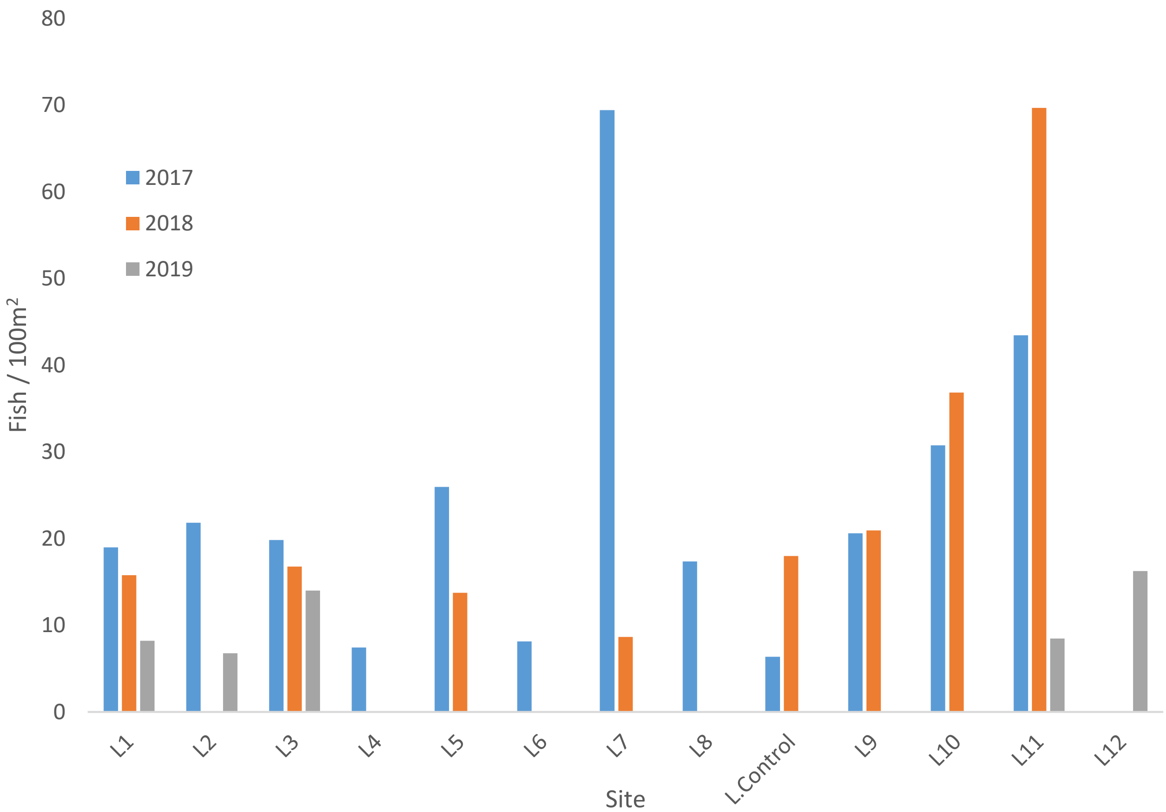


Figure B3. Calculated densities (minimum density estimate) of brown trout (older than fry) from Longcote 2017-2019. Note differing vertical axis to Figure B2.

Analysis

The results from the Kendall-Tau correlations are shown below in Table B2.

Table B2. Kendall-Tau results from Longcote burn showing correlation between time and catch

Site	d.f.	τ	p.
L1	5	-0.87	0.02
L2	5	-0.87	0.02
L3	5	-1	ns
L4	5	-0.73	ns
L5	5	-0.55	ns
L6	5	-0.73	ns
L7	5	-0.73	ns
L8	5	-0.73	ns
L Control	N/A	N/A	N/A
L9	5	-0.73	ns
L10	5	-0.69	ns
L11	5	-0.73	ns
L12	5	-0.73	ns

The results show a significant negative correlation between time and trout fry in two lower sites (L1 and L2), but not at any other location. However, it should be noted that all the other sites have a tendency towards a decline in fish numbers of the period of the study (as evidenced by the negative τ-values).

B4. Discussion

The density of the dominant brown trout fry class in the Longcote burn appears to be on a downward trajectory within areas that have been excluded from grazing; however it has not been possible to test whether this is occurring only within these excluded areas, or is a potential decline seen throughout the burn. The single partial control site that was available (L Control) does mirror the decline seen in other sites from the period 2017-2019, so it is possible that a wider reduction in fry productivity is occurring.

Another pertinent observation is that Kendall-Tau only really tests for linear direction of travel, rather then possible non-linear interactions in time. Therefore, annual factors (e.g. – seasonal flows) may have different effects on the number of fish observed (both positively and negatively) depending on year, which will not be picked up by the tests used here.

Small channels such as the Longcote are incredibly susceptible to singular events which can impact entire year classes of fish. Surveys in 2018 were potentially impacted by works carried out at the bottom of the Longcote during autumn and winter 2017/2018. A new footbridge crossing was built, with a pipe culvert used during construction to permit flows to pass. This culvert however collapsed and may have potentially prevented adult salmonids (and other fish species) from migrating into the Longcote in time for spawning. This potentially may have driven the decline observed from 2017 – 2018. However, at almost all sites, 2018 fry values were greater than 2019, when free access to the channel for migrating adult trout was available (albeit during a dry year – see main text).

In summary the sampling design limits the conclusions that can be drawn from the study; however, there is an observed decline in fish abundance seen in lower (downstream) sites, with a potential trend towards a reduction in others. This trend may be exacerbated by the susceptibility of the Longcote to both natural and anthropogenic events preventing access to the channel during key life-history periods such as spawning, and there is no evidence that declines are being caused by tree planting and exclusion with tentative evidence suggesting declines are co-occurring in channel reaches with grazed areas of bank.

Appendix C – Shiplaw Burn

C1. Introduction

The Shiplaw Burn is a small second order tributary of the Eddleston Water measuring 3.5 km in length. From 2013 to spring 2013, livestock fences were complimented by tree planting to create buffered exclusion zones on the burn as part of the Eddleston Water Project. As with the Longcote Burn, the Tweed foundation begun surveying the river in 2013 to establish the potential impacts of the exclusion on the fish community, dominated by brown trout fry (The Tweed Foundation, 2014b, 2015b & 2016b). These were carried out in 2013 – 2016. As part of Forth Rivers Trust contract to monitor fish populations in the catchment the sites were again monitored from 2017-2019.

C2. Method

Survey

The sites surveyed are shown overleaf in Figure C1. There was a total of 10 sites surveyed from 2013-2015 and 2017-2019, with a control, unfenced sample, taken at a site from 2016-2019. In a different approach to that on the Longcote, the length of watercourse at each site was standardised to 15 m and the time spent electro-fishing recorded to calculate the catch per unit effort. The rationale for doing this was to reduce the variation in survey length which could contribute to the variability of results (The Tweed Foundation 2013b). Surveys were carried out from 2013-2015 by the Tweed Foundation, with surveys in 2017, 2018 and 2019 carried out by a combined project team (Forth Rivers Trust and Tweed Foundation).

The site was identified and 15m of channel length measured out with tape. The site was then fished in an upstream fashion until the upper limit was reached, when fishing ceased. Fish were captured using handsets or using a banner net where the site was wide enough. The time taken to fish the site was recorded and total area fished calculated. Captured fish were placed in a bucket and processed using standard approaches (see main text), namely subdued with anaesthetic, speciated, measured and weighed, and returned to the site upon completion site survey.

Basic physical characteristics of the watercourse were recorded on the SFCC data recording form, including site lengths and widths, permitting the determination of site areas and thus a minimum density estimate of fish caught. Other features such as flow type and substrate, as well as riparian features were recorded.

Data description and analysis

Estimated fish density of each life history stage and species was calculated using measured area and catch standardised to 100m^2. However, total area fished was often a very small fraction of 100m^2 which led to a very large increase from actual count value to density value. This also potentially artificially amplified the sampler/sampling error and also the inherent variability in the system, so further analysis was carried out using abundance data only. Salmon were absent, while the concerns relating to the use of fish older than fry described within the main text also existed on Eddleston tributaries. Therefore, only brown trout fry abundance was used as a response variable.

Data from 2013-2015 was provided by The Tweed Foundation and added to the overall dataset. A time series analysis on each site was carried out using the conservative Kendall-tau correlation test to control for non-normality (Quinn & Keogh, 2002). Tests were carried out in R using Package “Kendall” (McLeod 2015) using the Kendall() function. To avoid spurious comparisons between the larger treatment dataset and the shorter control dataset, the latter was removed from further analysis.

Figure C1. Longcote electrofishing sites

C3. Results

Survey results

No salmon were recorded during the study period (2017-2019) and one large eel (350mm) was recorded at S4 in 2017. All remaining catch were brown trout.

The numbers of fish caught of the 2016-2019 surveys are shown below in Table C1, while the calculated densities are presented in Figure C2 and C3.

Table C1. Catch of brown trout fry and parr and older ("Older") from the Longcote 2017-2019

Site	2017		2018		2019	
	Fry	Older	Fry	Older	Fry	Older
S1	7	4	7	1	5	0
S2	5	8	4	1	1	2
S3	1	7	10	0	3	1
S4	0	3	7	0	1	0
S5	1	2	7	0	0	0
S6	3	4	12	2	0	1
S. Control	1	1	6	0	0	0
S7	1	4	12	0	2	1
S8	0	1	6	0	0	0
S9	2	0	6	0	0	0
S10	2	0	7	0	1	0

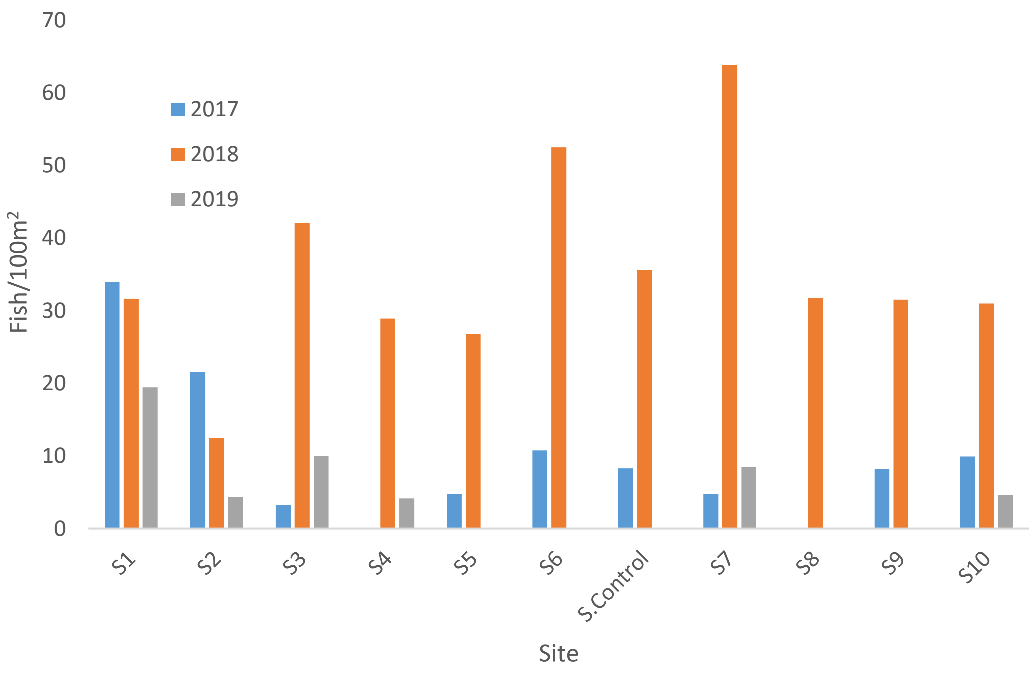


Figure C2. Calculated densities (minimum density estimate) of brown trout fry from Shiplaw 2017-2019

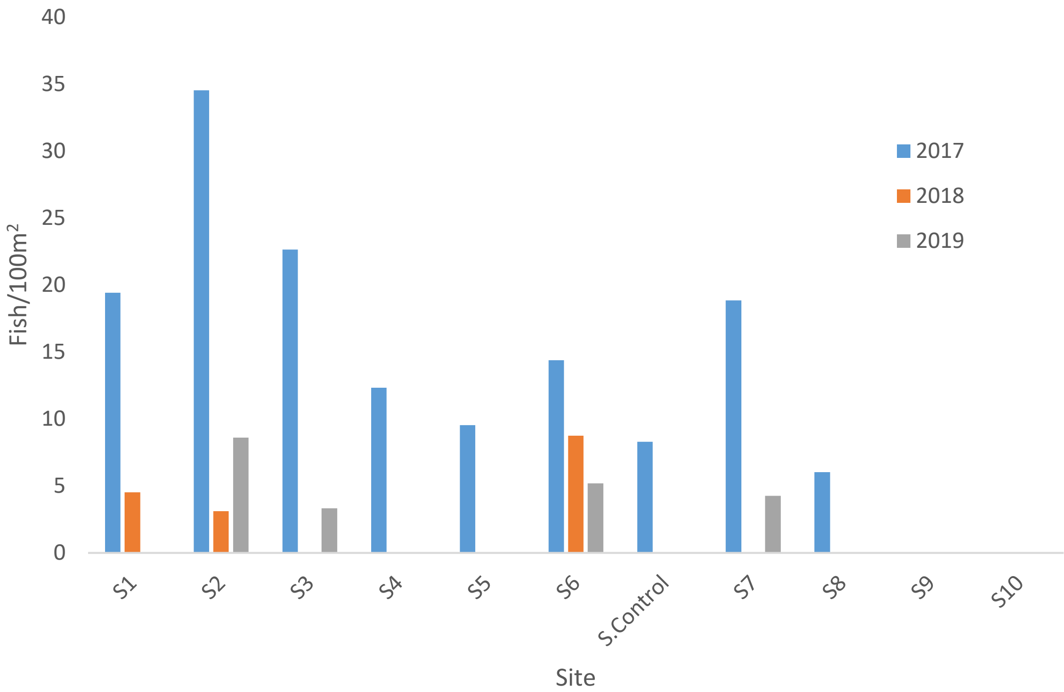


Figure C3. Calculated densities (minimum density estimate) of brown trout (older than fry) from Longcote 2017-2019. Note differing horizontal axis to Figure C2.

Analysis

In common with the rest of the study, only fry were used for detailed analysis ensuring a reasonably high confidence that the results reflect fish interaction with site over its entire lifespan. The results from the Kendall-Tau correlations are shown below in Table C2.

Table C2. Kendall-Tau results from Longcote burn

Site	d.f.	τ	p.
S1	5	-0.27	ns
S2	5	-0.55	ns
S3	5	-0.07	ns
S4	5	-0.47	ns
S5	5	-0.27	ns
S6	5	-0.33	ns
S Control	5	N/A	N/A
S7	5	-0.27	ns
S8	N/A	-0.27	ns
S9	5	-0.27	ns
S10	5	0.149	ns

The results show no significant reductions or increases in fish numbers at the sites over time; however, there was once again a general tendency towards a reduction in site number as evidenced by the negative τ – values. One site (S10) did show a tendency towards an increase

C4. Discussion

There does appear to be some suggestion that trout fry numbers may be in decline on the excluded Shiplaw; however 2018 in particular highlighted that numbers in some years numbers can be higher. This once again indicates that interannual variation in spawning and fry production may be large, possibly driven by flow levels impacting access to the burn. It has not been possible to test whether any observed patterns occur only within the excluded areas, but the one partial control that is available does show a commonality with the treatment sites.

Abundance of trout fry was also potentially impacted by the presence of a weir low down on the burn. This was mitigated in time for the migration period relate to the 2019 surveys. The original weir is shown in Figure C4, with the pass shown in Figure C5.

Figure C5a. Weir at the bottom of the Shiplaw Burn

Figure C5b. Weir with wooden fish pass installed pre-autumn 2018

This weir will have amplified any low flow effects on adult access to the Shiplaw for reproduction. The results from 2018 particularly suggest that any potential changes over time are non-linear, and the inability of Kendall-Tau to detect non-linear trends should be considered (See Appendix B).

Again, the sampling design limits the conclusions that can be drawn, however there was no definitive negative (or positive) correlation between time and fish abundance. It is possible that interannual fluctuations driven by both natural (seasonal climactic conditions) and anthropogenic (the presence of a weir and subsequent mitigation) are the dominant drivers of fish populations on the Shiplaw. There is no evidence that changes are being caused by tree planting and exclusion.