THE UNIVERSITY OF HULL

IMPLICATIONS OF SMALL-SCALE RUN-OF-RIVER HYDROPOWER SCHEMES ON FISH POPULATIONS IN SCOTTISH STREAMS

being a Thesis submitted for the Degree of

Doctor of Philosophy

in the University of Hull

by

Andrea Lauren Robson

B.Sc (Hons)(Hull)

August 2013

CONTENTS

List of Tables

List of Figures

Acknowledgements

First and foremost I offer my sincerest gratitude to my supervisor, Dr Jon Harvey, who has supported me throughout my thesis. His patience and kindness, as well as his academic experience, have been invaluable to me; I could not have wished for a better or friendlier supervisor. I would also like to thank Professor Ian Cowx, without him I would not have been given this incredible opportunity to study for a doctorate. I am truly grateful for his encouragement and support.

I have been surrounded by an incredible team at Hull International Fisheries Institute where I carried out my study. I gratefully acknowledge the assistance of Dr Jon Harvey, Dr Andy Nunn, Dr Richard Noble, Dr Jonathan Bolland, Dr Ryan Taylor and Mr Sam Walton during fishing surveys in Scotland. I would also like to thank external organisations SEPA, Lochaber Fisheries Trust and Morgan Fisheries Consultancy for allowing me access and use of their data sets within my study.

Finally, I thank my family and friends for supporting me throughout all my studies at university. My parents especially, Peter and Christine Robson, have provided a constant source of support – emotional, moral and, of course, financial.

Abstract

In the past few years there has been a resurgence of interest in hydropower as a direct consequence of the UK government's commitment to renewable energy and associated financial incentives. The majority of new schemes are run-of-river, which have no significant storage of water, the turbine only making use of the available flow at the site. Hydropower is often presented as a clean and renewable energy, and thus portrayed as having no negative impacts on the environment. However this description has been challenged by numerous authors who consider the impacts on fish and other biota as significant, particularly on salmonid fish populations in relation to migration.

This study investigated the implications of small-scale run-of-river hydropower schemes on fish populations in Scottish streams. In these schemes, water is abstracted from an intake above a mall weir to drive a turbine before the water is returned to the watercourse at a downstream location. Abstracted water is channelled down a pipeline to the turbines in a powerhouse before release at the outfall position; this results in a depleted reach. The term "depleted reach" refers to the stretch of river between the intake and outfall of high-head run-of-river hydropower schemes that experiences reduced flow due to abstraction. The main impact in the depleted reach is a reduction in the amount of water, leading to associated changes in habitat including important spawning/nursery areas. The main impact upstream of the intake is reduced access because of the intake weir, which may be exacerbated by the reduction in the amount of water downstream. Therefore, impacts can be observed upstream of an intake (barrier effect), upstream and downstream of an intake (barrier and abstraction effect) and downstream of an intake (abstraction effect). In total, ten schemes were included within this study; five with extensive pre-and post-monitoring and a further five that were considered to have less extensive data. At Kinnaird Burn, Keltney Burn and Innerhadden Burn, salmonid populations varied over the study period. Densities of fish varied both within and outside the depleted reaches, therefore, the inter annual variations in salmonid densities made it difficult to detect any impacts, specifically in response to commissioning of the hydropower schemes, when comparing before/after and control/impact data, despite having extensive pre- and post-commissioning data. It was difficult to detect any impacts of the Ardvorlich Burn, Douglas Water, Camserney Burn and Allt Gleann Da-Eig hydropower schemes on fish densities due to the limitations of the data sets, including a lack of baseline and post-commissioning data and control sites to account for temporal and spatial variations in the fish populations. Consequently, confident conclusions could not be drawn. In the River Callop, 0+ salmonid densities declined at several sites in the depleted reach post hydropower commissioning. However, a lack of spatial and temporal data made it difficult to conclude whether the decline was in response to the hydropower scheme or natural variability. At Rottal Burn and Inverhaggernie Burn, a reduction in ≥1+ and 0+ salmon density respectively, was observed in the depleted reach, post hydropower commissioning. These declines were not reflected in the fish densities at the control site downstream of the depleted reach and thus suggest an impact of flow regulation.

The meta-analysis of historical data and subsequent monitoring raised issues about the Environmental Impact Assessment strategies on some of the schemes. Therefore the concerns that existing sampling protocols and impact assessments are inadequate to provide robust, defensible information about the impact of small-scale run-of-river hydropower schemes on fisheries, was upheld. A proposed survey protocol was developed using Before After Control Impact analysis that is intended to answer this need and to be used in conjunction with appropriate guidance documents provided by the regulatory agencies, such as the Scottish Environment Protection Agency (2010) "*Guidance for developers of run-of-river hydropower schemes*" and the Environment Agency (2009a) "*Good practice guidelines to the Environment Agency hydropower handbook*".

CHAPTER ONE

1 GENERAL INTRODUCTION

Rivers provide an array of ecosystem goods and services, including provision of habitat for biodiversity, attenuation of flood waters, abstraction, recreation, production of power, food and other marketable goods (Postel & Carpenter, 1997; Millennium Ecosystem Assessment, 2005), the latter based on artisanal and commercial fisheries, aquaculture, floodplain regression agriculture and pastoral animal husbandry (Postel, 2005; Welcomme *et al.,* 2006; Sala *et al.,* 2008). Consequently rivers have been widely altered by a suite of interacting human actions, including effluent discharge, dam building, habitat alteration and water abstraction (Baron *et al.,* 2002; Nilsson *et al.,* 2005). These actions can drastically alter the ecological structure and ecosystem functioning, and have resulted in aquatic organisms being among the most threatened species groups in the world (IUCN, 2008).

Climate change is one of the greatest environmental challenges facing humankind (Jonsson & Jonsson, 2011) and with concerns over this phenomenon escalating, rivers worldwide are becoming increasingly subject to various levels of physical alteration and river regulation to provide humans with services such as hydropower and water abstraction (Jansson, 2002; Murchie *et al.,* 2008). These issues threaten freshwater fish populations by reducing spawning areas and nursery habitats, and altering flows, sediment and temperature downstream of constructions (Anderson *et al.,* 2006; de Leaniz, 2008; Lucas *et al.,* 2009). Although the harnessing of energy from water discharge and conversion to electrical power did not begin until the mid-19th Century (Poff & Hart, 2002), hydropower is now considered the most important renewable electricity source worldwide (Bratrich *et al.,* 2004), accounting for 19% of the world's electricity (Paish, 2002). Furthermore, the International Energy Agency (IEA) predicts that hydropower output worldwide will increase from 2809 TWh in 2004 to 4749 TWh by 2030 (Institution of Engineering and Technology, 2007). Hydropower was considered to be the most reliable and cost effective renewable energy source (Bruno & Fried, 2008) in the UK for almost 10 years between 1997 and 2006, being only recently overtaken by wind power in 2007 (DECC, 2010). Hydropower is often presented as a clean (Rosenberg *et al.,* 1995) and renewable energy source that is environmentally preferable to fossil fuels or nuclear power (Renofault *et al.,* 2010). However, this energy does come with major economic, social and environmental limitations (Demirbas, 2007). A general misconception of hydropower is that it remains a clean source of energy, as there is no carbon dioxide, sulphur dioxide, nitrous oxides or any other emissions and no solid or liquid waste production (Ramos & Almeida, 1999). It could be argued that the construction process may involve release of some toxins but this is unknown. Hydropower is therefore often portrayed as having no negative impacts on the aquatic environment, but this description has been challenged by authors, including Ausubel (2007) who consider the impacts on fish and other biota as significant.

1.1 Types of hydropower

Hydropower schemes can be classified in two categories; “impoundment”, which are generally large schemes, and “run-of-river”, which are generally smaller. Small-scale hydro electricity is generally used to describe all hydropower plants with a production capacity of less than 10 MW (Larinier, 2008). Run-of-river hydro electric schemes can be of three main types. Low-head schemes are those that use a head of water between 5-25 m. A head of 25-50 m is classified as “medium-head” schemes and those of 50 m and above are “high-head” schemes. The head of water relates to the vertical distance from the intake at the top of the scheme to the floor level of the turbine at the bottom of the scheme, at the outfall (McKenzie, 2007). Generally, the higher the head developed, the greater the power output (Hogan, 2005).

Run-of-river schemes are considered to be more environmentally friendly Fraenkel *et al.*, 1991), as the social and environmental impacts are suggested to be much less severe, than traditional large impoundment hydropower schemes. Irrespective of this there still generally remains very little empirical data on the ecological effects and response of fish communities to run-of-river hydropower schemes (REFOCUS, 2002; Habit *et al.,* 2007). Consequently these smaller developments are poorly understood and not adequately studied (March *et al.,* 2003).

There are well documented impacts of large impoundment hydropower schemes, with changes to fish population structure, habitat alterations, loss of crucial spawning and nursery habitat, loss of biological diversity, modifications to water quality and hydrological regimes, and disruption of longitudinal connectivity being some of the key factors threatening fish populations (Bunt *et al.,* 1999; Marchetti & Moyle, 2001; Trussart *et al.,* 2002, Robertson *et al.,* 2004; O’Connor *et al.,* 2006; Fette *et al.,* 2007; Habit *et al.,* 2007; Poulet, 2007; Murchie *et al.,* 2008; Lucas *et al.,* 2009). However, while many of these impacts are manifest on large impoundment hydropower schemes, the issues are equally relevant to small-scale run-of-river hydropower schemes (Table 1.1).

Table 1.1 Comparison of key characteristics of low-head run-of-river, high-head run-of-river and large impoundment hydropower schemes.

Characteristic	Low-head run-of-river schemes	High-head run-of-river schemes	Large impoundment schemes
Location in catchment	Generally in lowland reaches, often constructed around existing weirs	Upland reaches associated with waterfalls or steep gradient terrain	Mostly upland catchments, impounding steep valleys
Impoundment types	Generally use existing weir structures >3 m high.	Small weir <3 m often passable at high flows. May be constructed on existing impassable waterfall.	Large dam “usually” or “may be” impassable to fish.
Hydrological features	Diversion of flow, separate channel housing turbine leading to depleted reaches between impoundment and outfall, except when no abstraction, or diversion of flow through turbine constructed on impounding structure Operation except at high or low flows.	Diversion of flow through turbine leading to depleted reaches between impoundment and outfall, except when no abstraction. Operation except at high or low flows.	Water storage and release, linked to electricity demand. Can lead to hydropeaking of regulated flow. Hydrology upstream of impoundment converted from lotic to lentic system.
Fish community characteristics	Cyprinid or salmonid-dominated – may have migratory species attempting to traverse weir.	Salmonid dominated. Often located at or upstream of migratory salmonid limits.	Salmonid dominated prior to construction, but shift upstream from lotic to lentic species following construction.
Reservoirs	No	No	Yes
Mitigation	Existing fish passes may be present on weir – fish passes may be required under abstraction licence Hands-off flows Screened intakes/outfalls	Small V-notch weirs may not have fish passes. Larger weirs may have fish passes. Hands-off flows Screened intakes/outfalls	Fish ladders/lifts, often species specific. Environmental flow releases Screened intakes/outfalls

1.2 Small-scale hydropower potential in the UK

In the UK, there has been a resurgence of interest in hydropower in the past few years as a direct consequence of the government’s commitment to renewable energy and the associated financial incentives, especially for small-scale schemes (<10 MW installed capacity). The introduction, from April 2010, of a Feed-In Tariff – guaranteeing a high fixed price for the electricity generated for up to 20 years – is likely to encourage further small hydropower projects.

Scotland, Wales and upland England are considered to be of high interest for hydropower development because of their mountainous topography and temperate maritime climate (Copestake, 2005). Hence, seasonal differences in temperature and rainfall are not severe and rain is present throughout the year; irrespective of this, it must be noted that rivers in Scotland still experience low flows during the summer. Scottish rivers therefore have abundant flow volumes and high water quality (Gilvear *et al.,* 2002). In addition, there are many more opportunities for low-head schemes to be retrofitted on the numerous weirs and obstructions in lowland rivers throughout England and Wales. Indeed, the Environment Agency (EA) mapped the hydropower opportunities in England and Wales (Figure 1.1; EA, 2010a) and identified 25,935 obstructions as potential candidates for development. A similar study in Scotland (Figure 1.2; Forrest *et al.,* 2008) identified 36,252 sites for potential development. Potential hydropower structures were based on in-river features, and include waterfalls, weirs, dams, barrages and locks. However, even if all suitable sites are developed for hydropower, the theoretical unconstrained maximum energy that could be generated (1100 MW) is equivalent to only ~1% of the UK's requirements (H. Huyton, Environment Agency Climate Change Policy Advisor, pers. comm.).

Figure 1.1 Location of obstructions as potential candidates for hydropower development in England and Wales (source EA, 2010a).

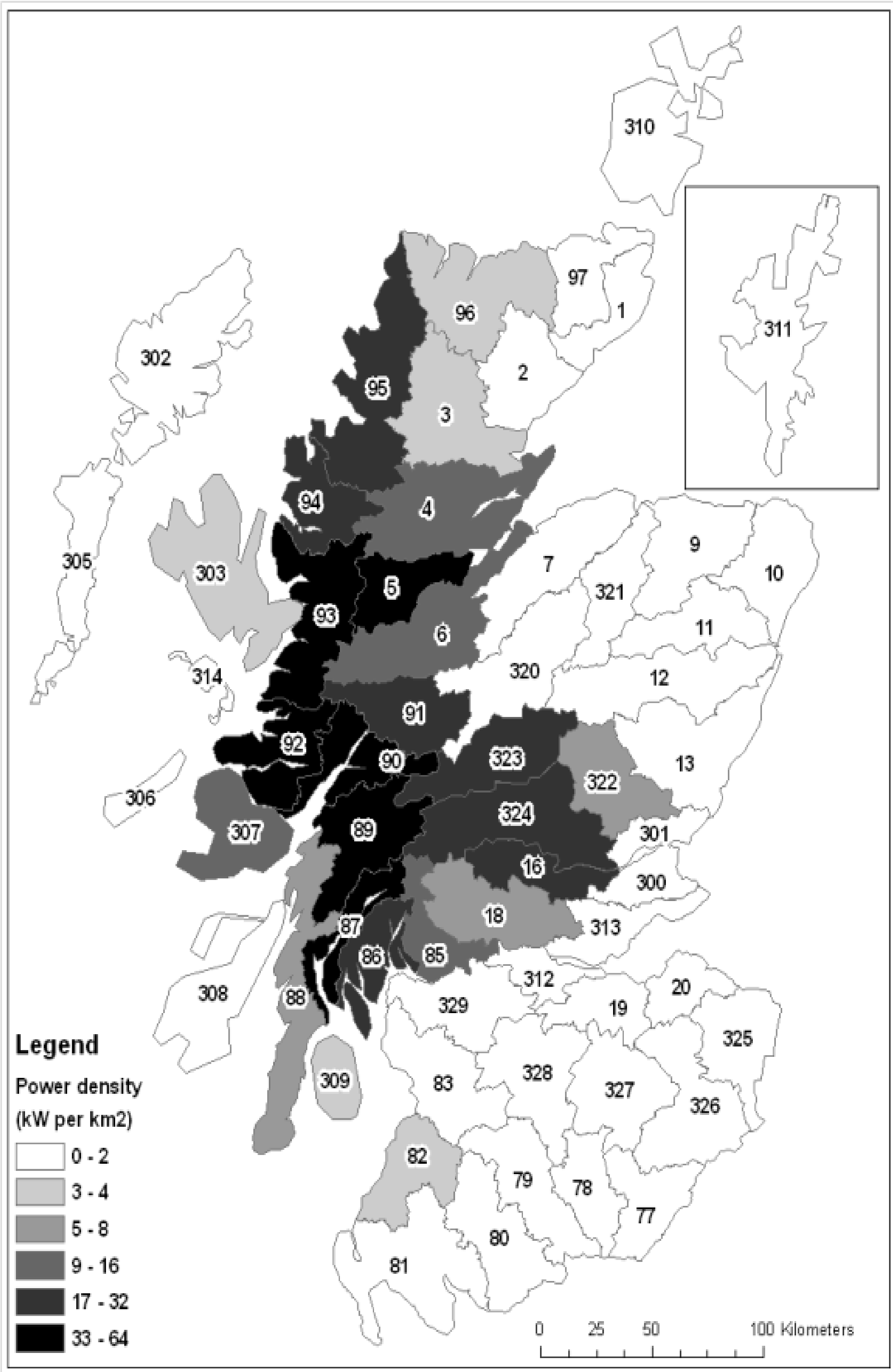

Figure 1.2 Map of Scotland divided into 60 catchments. Numbers within catchment refer to catchment ID ranging from 1-329. Shading indicates hydropower potential within that catchment, divided by the area of the catchment, to give power density (source Forum for Renewable Energy Developments in Scotland [FREDS]; Forrest *et al*., 2008).

In many cases, the most potentially viable hydropower sites are unfortunately on the more ecologically important rivers. Hydroelectric generation in 2006 was the highest volumetric use of water in Scotland (MacLeod *et al*., 2006). In 2003, only 10% of electricity generated in Scotland was produced by renewable sources. The Scottish Executive reported that 18% of electricity consumed should come from renewable generation by 2010, and should have risen to 40% by 2020. Due to the topography of Scotland, with high gradient and torrential character, the land is perfect for accommodating small-scale hydropower schemes (Copestake, 2005). Catchments with hydropower schemes cover 20% of the area contained within mainland Scotland (Gilvear *et al.,* 2002). Additionally, a high number of them in Scotland are now associated with either dams, impoundments, or water transfers (Gowans *et al.*, 2003).

1.3 Study species

The species that are likely to be impacted by the run-of-river hydropower schemes in this study are Atlantic salmon (*Salmo salar* L.), brown trout (*Salmo trutta* L.) and European eel (*Anguilla anguilla* (L.)). Each of the species has a complex life cycle, which has been described in detail by many authors (Grassi, 1896; Schmidt, 1922; Crisp, 2000; Klemetsen *et al.,* 2003; Shiao *et al.,* 2006; Jonsson & Jonsson, 2011; Righton *et al.,* 2012); the following section provides an overview of the life cycle of the Atlantic salmon, brown trout and European eel.

1.3.1 Life histories of salmonids

Careless human attitudes and actions in the past have exterminated many salmonid populations and dramatically weakened others (Jonsson & Jonsson, 2011); they are continuously facing new challenges threatening their populations. To some degree salmonid populations are controlled by the abiotic factors of geomorphology, water quality and quantity and hydrology. They stand as symbols of clean cold water (Jonsson & Jonsson, 2011) and so are great indicators of a river's ecological status (Gilvear *et al.,* 2002). Environmental factors such as water temperature, flow, depth, substratum composition, nutrient richness and habitat coherence and consistency are all drivers of a salmonid's life cycle and it's demographic traits; as such these factors can all influence it's life history traits (Jonsson & Jonsson, 2011). Both the Atlantic salmon and brown trout spawn in freshwater in autumn and winter and thereby differ from most other fresh water fish in the UK, which are spring or summer spawners. Both species' preferred spawning areas are swift-flowing rivers and streams; those found abundantly across Scotland.

1.3.1.1 Atlantic salmon

The Atlantic salmon is undoubtedly one of the most studied freshwater species in the world and exists in both anadromous and non-anadromous forms in river systems on both sides of the North Atlantic Ocean (Klemetsen *et al.,* 2003). Anadromous fish are those that return to their natal spawning rivers from the sea to breed. The UK Atlantic salmon population comprises a significant proportion of the total European stock. Scottish rivers in particular are a European stronghold for the species as approximately 80% of the UK resource is found in Scotland. Overall, the UK Atlantic salmon populations are considered to be in decline, with approximately 80% of the resource considered in a stable condition (i.e. the Scottish population). However, whilst the overall Scottish rod catch figures have remained reasonably stable since 1994, this is not the case for all stock components; early running multi-sea-winter fish, for example, have declined significantly in many rivers and this continues to be a cause for concern (Joint Nature Conservation Committee, 2007). Many publications have discussed the rapid decline in salmon populations (Jonsson & Waples., 1999; Prevost & Chaput., 2001; Gilvear *et al.,* 2002), partly due to the construction of barriers to migration (Lucas & Baras, 2001). The reality is however, despite an abundance of publications providing immense information, that the complexity of the salmon life history still leaves unanswered questions.

The Atlantic salmon life cycle (Figure 1.3) is very complex and requires precise environmental conditions at each life stage.

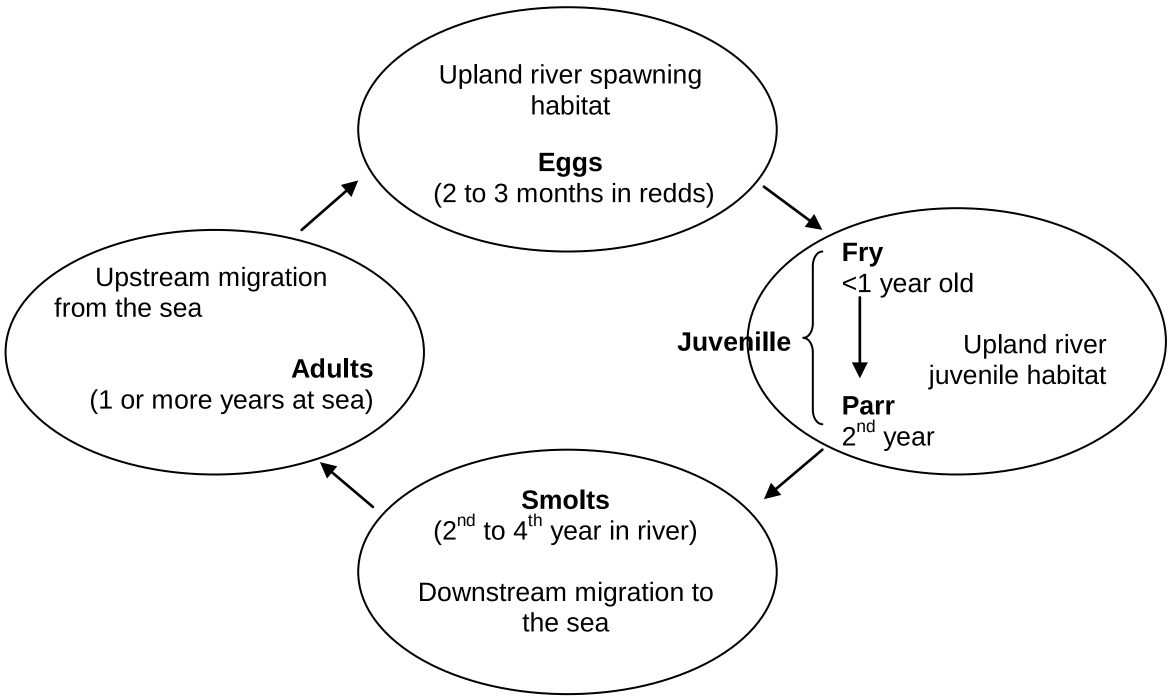

Figure 1.3 Life cycle of Atlantic salmon (Noble *et al*., 2009).

The cycle begins as small reddish orange eggs, which are hidden in vast quantities under the gravel in fast-flowing streams, ensuring protection from predators. Embryos develop during the winter (Jonsson & Jonsson, 2011), and at the end of the incubation period in the subsequent spring, the eggs hatch and are referred to as alevins. At this stage, alevins are extremely vulnerable, they are small and weak and unable to swim, making them an easy target for predators. Due to this, alevins remain under the gravel with only their yolk sac, which must last for several weeks to provide essential nutrients (Berg *et al*., 2001). Once the entire yolk has been absorbed, which can take up to six weeks, the alevins emerge from the gravel as fry (Crisp, 2000), at approximately 2 cm in length (Mills, 1989). During the “swim-up” stage, the fry fill their swim bladder with air to attain neutral buoyancy. After this fry are able to leave the gravel to forage (Crisp, 2000). Once fry have developed and grown to around 5 cm long, they are referred to as “parr” (Crisp, 2000); parr are still vulnerable to predators at this stage due to their small size.

Parr can remain in fresh water for between 1 and 4 years (Malcolm *et al.,* 2010), where they feed on aquatic insects. At the end of the parr stage, fish become smolts prior to their migration to sea. At this stage salmon undergo a physiological transformation called smolting, which is a pre adaptation for marine life (Jonsson & Jonsson, 2011) to enable survival in the sea. This is the most crucial stage and mortality is high. The age at which parr undergo smolting is suggested to be related to their size; smolting usually occurs at lengths of 12.5-17.0 cm (Crisp, 2000). Smolts then move out to their feeding grounds between March and June, where they live in schools and feed excessively to store fat and energy. Post-smolts of Atlantic salmon feed in the north Atlantic and can migrate more than 2000 km in the open ocean away from their home river (Jonsson & Jonsson, 2011); adult salmon from Scottish rivers pass through or make use of areas around to the west and east of Greenland and the Faroe Islands (Malcolm *et al*., 2010). The length of time salmon spend at sea varies between 1 (grilse) and 4 (Multi Sea Winter salmon) winters, before attaining maturity and returning to their natal river to spawn. Although it is considered salmon attain sexual maturity during time at sea, the importance of precocious parr should be noted. These male parr are morphologically juvenile yet sexually mature and capable of fertilizing a large proportion of eggs (Garcia-Vazquez *et al.,* 2001). Precocious parr therefore help maintain genetic variation in wild populations, which is essential for conservation genetics.

Upon reaching the spawning grounds, generally located in the upper reaches of rivers, females dig nests called redds, which are hollows in gravels (Gilvear *et al.,* 2002). The substrates available for spawning salmon have been classed as good (40-80% gravel;

10-40% cobble; <20% boulder; <20% combined silt and sand) and marginal (<40% gravel; 50-90% combined gravel and cobble; <20% combined silt and sand) (Semple, 1991). Investigations have reported the range of substrate sizes used by Atlantic salmon to generally be 16-64 mm, however, this can differ depending on the size of river with salmon in large rivers using coarser substrata (32-128 mm) compared to those in small rivers (16-32 mm) (Louhi *et al.,* 2008). Redds observed in Scottish burns were recorded at 2-3 m long, 1-1.5 m wide and 0.2-0.3 m deep (Moir *et al.,* 1998). While the amount of eggs a female lays can be dependent on her size, studies have shown this number can be between 2,000 and 15,000 per female (Mills, 1989). Males follow the females covering the eggs with milt/sperm, which are later covered by gravel. Salmon usually bury their eggs 15-25 cm deep (Armstrong *et al.,* 2003) thus a habitat comprised of clean gravel in abundance is of great importance. After spawning the adult salmon are known as kelts. Atlantic salmon are iteroparous meaning that many survive reproduction and can spawn repeatedly (Schaffer, 1974; Jonsson & Jonsson, 2011). Adult Atlantic salmon will travel back to sea spending 12-15 months feeding, before returning to the river to repeat the spawning cycle. Large Atlantic salmon typically wait two years before spawning again; few individuals spawn more than three times and live longer than 10 years (Jonsson & Jonsson, 2011).

1.3.1.2 Brown trout

Brown trout is one of the few native species found in Scotland and is a UK Biodiversity Action Plan (UK BAP) Priority Species (Malcolm *et al.,* 2010), exhibiting one of the most diverse life histories (Figure 1.4) amongst fish in the world (Jonsson, 1989).

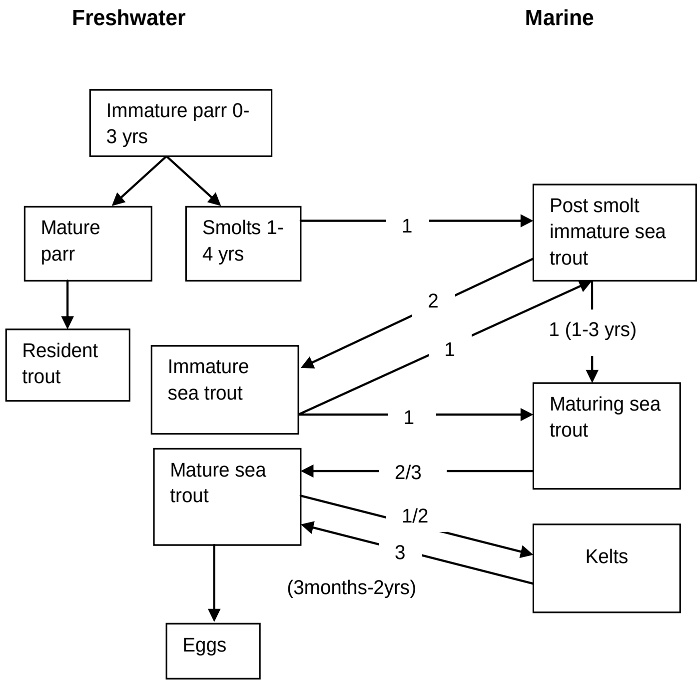

Figure 1.4 Life cycle of sea trout. (Celtic Sea Trout Project, 2010). 1 = feeding; 2 = wintering; 3 = spawning.

Brown trout exploit an array of localities from maintaining residential positions to performing migrations between spawning, nursery and feeding areas (Jonsson & Jonsson, 2011), thus are one of the most adaptive fish in northern waters; taking both anadromous and non anadromous forms. Anadromous brown trout are referred to as sea trout; they migrate to coastal areas to feed before returning to fresh water to spawn. Immature sea trout, however, often return to fresh water to over-winter (Malcolm *et al.,* 2010). Migration duration varies amongst populations and while distances of up to 2000 km have been reported (Jonsson *et al.,* 1994*;* Jonsson & Jonsson, 2006) these are considered exceptional circumstances. Nonetheless brown trout exhibit a wide range of life history strategies. Studies have revealed that on the west coast of Scotland, many sea trout may use locally constrained areas, migrating to nearby coastal areas. On the east coast, however, fish exhibit wide ranging migrations suggesting that offshore movement and migrations is also a feature of sea trout behaviour (Malcolm *et al.,* 2010). Brown trout that occur in rivers or lakes with access from the sea often form anadromous populations, although females are more inclined than males to become anadromous (Klemetson *et al.,* 2003). Non anadromous brown trout are referred to as resident brown trout; they do not migrate to sea but instead remain in fresh water for their entire life. While both sexes of brown trout frequently mature and become freshwater resident instead of migrating to sea, they still display seasonal movements and migrate among different habitats, each serving different purposes such as feeding, spawning and wintering. Populations are often partially migratory meaning one part of the population will leave and feed elsewhere, while another part stay as residents (Klemetson *et al.,* 2003). Brown trout and the anadromous sea trout are commonly regarded as constituting a single species (Campbell, 1977).

Brown trout inhabit the same or nearby rivers as salmon with similar environmental conditions (Jonsson & Jonsson, 2011) but given suitable spawning substratum, correct temperature conditions and sufficiently good water quality, brown trout will occupy habitats from very small streams to the largest rivers (Klemetson *et al.,* 2003). Trout also spawn during autumn or early winter but when living sympatrically with Atlantic salmon brown trout often spawn earlier in the autumn (Jonsson & Jonsson, 2011). Differences between spawning periods have also been observed between sea trout and brown trout; the former from early November to the first week of December, and from the middle of October to the third week of December for the latter (Campbell, 1977). Trout spawn on stone (not bedrock) and gravel bottoms, usually in running waters (Klemetson *et al.,* 2003). Mirroring that of salmon, females dig nests within the gravel called redds, within which the eggs are submerged and protected. Substrate sizes used by brown trout are broadly in the gravel range 8-128 mm (Armstrong *et al.,* 2003) although large females often spawn on coarser gravel and bury their eggs deeper than small ones (Fleming, 1996). Females spawn in two portions, the second portion of eggs can be placed in a nest directly in front of the first nest, or eggs could be spawned elsewhere in the river (Klemetson *et al.,* 2003). One female is often courted by several competing males (Klemetson *et al.,* 2003) with brown trout males participating in the spawning activities of female sea trout (Campbell, 1977).

The life cycle of trout is similar to that of salmon with respect to the physical development stages of alevins and fry (see Section 1.3.1.1) but after this the ecological variability of trout (Klemetson *et al.,* 2003) allows them to occupy different habitats when living sympatrically with salmon to avoid competition for food resources and habitat. As juvenile Atlantic salmon and brown trout commonly occur together in streams and rivers, it is believed that there is competition between the species (Stradmeyer *et al.,* 2008). However, when occupying the same reach of river, brown trout prefer deeper stream areas with moderate to low water velocities and rocky substrates, whereas young Atlantic salmon chose faster flowing and often shallower areas (Heggenes, 1996, 2002). While fry often stay in their natal stream for the first summer (Elliott, 1986), habitat use during the subsequent developmental stages (growth into parr) varies among populations (Klemetson *et al.,* 2003).

Klemetson *et al.* (2003) identified three different versions of the trout life cycle.

In the first, juveniles migrate from the natal river to a larger river and in the second young trout migrate from their natal stream into a lake. The third life-history type is found in anadromous trout (sea trout), where juveniles migrate to the estuary or coastal areas for feeding and return to fresh water for spawning or wintering. There is however a fourth in the form of resident brown trout that spend their entire lives in the same watercourse. Anadromous trout can remain as parr for a few years before they migrate to the sea to feed, however, sea trout post-smolts tend to use near shore sea loch and fjord areas where available rather than migrating rapidly out to sea from inshore coastal areas (Malcolm *et al.,* 2010). Time spent in the ocean may be for one summer only or for two or more years, before returning to their river of origin for spawning (Jonsson & Jonsson, 2002).

1.3.2 Life history of European eel

The European eel is a commercially valuable species (Shiao *et al.,* 2006) and currently listed as critically endangered on the IUCN Red List of threatened species (Freyhof &

Brooks, 2011). As the European eel forms a single stock that is distributed across the European continent, there is a Europe-wide issue. Recruitment of eels has declined throughout Europe since the early 1980s (Dekker, 2000). Specifically, recruitment of the glass eel stage of the European eel has fallen on average to 5% of the peak levels of the late 1970s and early 1980s (measured Europe-wide) (Dekker *et al*., 2007). A similar trend is noticed in the UK and according to government agency estimates and catch returns, they have declined by around 85% since the 1980s (Aprahamian *et al.,* 2007). International Council for Exploration of the Seas (ICES) advised that the stock is now outside safe biological limits (ICES, 2006). In response, the European Commission initiated an Eel Recovery Plan (Council Regulation No 1100/2007) to try to return the European eel stock to more sustainable levels of adult abundance and glass eel recruitment. According to EU legislation, EU countries need to take measures that allow 40 % of adult eels to escape from inland waters to the sea, where they can spawn. To demonstrate how they intend to meet the target, EU countries have drawn up national eel management plans at river-basin level. In December 2008, the UK submitted 15 eel Management Plans for individual assessment to the European Commission, covering the River Basin Districts, as defined under the Water Framework Directive, in England and Wales, Scotland and Northern Ireland. These plans aim to achieve an escapement of silver eel to the spawning population that equals or exceeds a target set at 40% of the potential biomass that would be produced under conditions with no anthropogenic disturbance due to fishing, water quality or barriers to migration.

All *Anguilla* species display a remarkable multi-stage life cycle (Chadwick *et al.,* 2007) and while the main features of the life cycle of the European eel (Figure 1.5) are well known there are still gaps concerning the oceanic life stages. This is mainly due to the difficulty of observing or obtaining samples of eels during their marine life stages (Bonhommeau *et al.,* 2010).

Eels are catadromous fish meaning they spend most of their lives in fresh water (on average 10-30 years; Prosek, 2010) and migrate to the sea to breed. Eels perform some of the longest known seaward migrations, with the European eel travelling more than 6000 km across the Atlantic Ocean (Bonhommeau *et al.,* 2010). For a long time the spawning of freshwater eels had been a mystery because no one could find their eggs or larvae around their familiar habitats such as rivers, ponds or coastal habitats (Tsukamoto, 2009). Together, Grassi (1896) and Schmidt (1922) closed the migratory loop of European eels. Grassi (1896) identified the larval form of eels and Schmidt (1912) later used this knowledge to find the densest aggregations of larvae to identify where the European eel spawned; far offshore in the Sargasso Sea of the Atlantic Ocean (Belpaire *et al.,* 2009; Bonhommeau *et al.,* 2010).

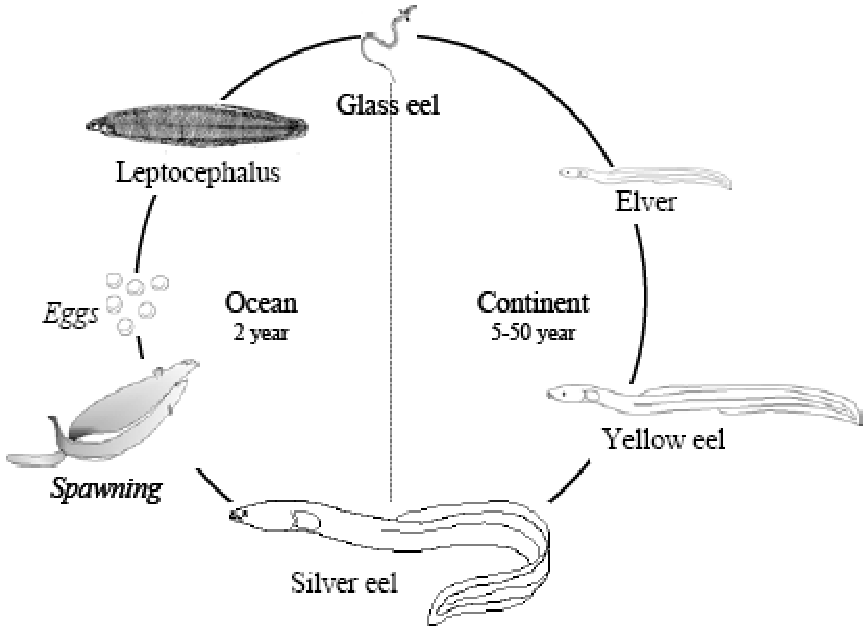

Figure 1.5 Life cycle of the European eel (OSPAR, 2010)

Conveyed by warm currents, eel larvae (leptocephali) drift from the spawning area in the Sargasso Sea towards the coast of Europe by the Gulf Stream. During this stage, eel larvae undergo metamorphosis into glass eels before reaching continental waters where they grow in fresh water and coastal habitats during their sedentary yellow eel phase (Belpaire *et al.,* 2009), where they can spend up to a century or more (Tesch, 2003). The length of time it takes European eel larvae to reach these areas has been very difficult to establish and different methods have led to different estimates of the duration of migration ranging from seven months to more than six years (Bonhommeau *et al.,* 2010). Prior to their migration across the Atlantic Ocean to spawning grounds in the south-western Sargasso Sea, yellow eels that live in rivers, lagoons or coastal waters mature into silver eels (Bonhommeau *et al.,* 2010). This silvering process begins weeks to months before migration (Fontaine *et al*., 1995) when eels undergo various morphological and physiological changes (Tsukamoto, 2009) that prepare them for their migration across the ocean. *Anguilla* species, however, do not undergo sexual maturation during the silvering process (Righton *et al.,* 2012); the natural process of sexual maturation remains almost unknown as the final part of *Anguilla* species lives is so difficult to study under natural conditions (Righton *et al.,* 2012).

As discussed, the rapid decline in eel population is well acknowledged with overfishing and anthropogenic activities including habitat loss and migration barriers (Belpaire *et* *al.,* 2009) reported as contributing factors (Dekker, 2003; Shiao *et al.,* 2006). However, the introduced parasite *Anguillicoloides crassus* (Kuwahara, Niimi & Itagaki) is also thought to play an important role in the decline of freshwater eel populations. The parasitic nematode was originally found throughout East Asia in its native host the Japanese eel (*Anguilla japonica* (Temminck & Schlegel)) but has been discovered to infect the European eel; introduced in the early 1980's via the live import of Japanese eels for the food trade (Kirk, 2000). The parasite compromises performance (Gollock *et al.,* 2005) of eels causing destruction of the glandular swim-bladder mucosa (Bernies *et al.,* 2011). This not only affects pressure balance but also causes increased energy costs. Investigations have shown that eels infected with nematodes have a swimming speed 18.6% slower than uninfected eels (Sprengel & Lüchtenberg, 1991). This is particularly worrying as it can affect the ability of silver eels to migrate in order to complete their life cycle (Kirk, 2003). Additionally, there are also possible effects of climate change causing future concern for eel populations. Due to the life cycle of the European eel, leptocephali depend solely on oceanic currents to transport them to European shores where they inhabit rivers for their adult life stage. Increases in water temperature, however, may weaken the Gulf Stream that runs north towards Europe transporting the eel larvae; the failure of this system could potentially inhibit the life cycle of the European eel. Indeed, the parallel decline in European and American eels, both of which spawn in the Sargasso Sea, has been taken as evidence that changes in ocean currents, resulting from climate change may have interfered with larval transport leading to reduced recruitment in both stocks (OSPAR, 2010). The discovery of the European eel's life cycle has also had repercussions in terms of fisheries management since eels fished or killed in fresh water have not yet had the chance to reproduce (Bonhommeau *et al.,* 2010), thus the protection and survival of eels is incredibly important.

1.4 Aims and objectives

Although the impacts of traditional large hydropower schemes on fish populations are well documented, the actual impacts of run-of-river hydropower schemes have not been adequately studied. While many of the principles from larger schemes can be applied to these smaller schemes it is critical to improve the understanding of the potential impacts of run-of-river hydropower schemes on fish populations, which will be achieved through this study. There are a number of operational and proposed run-of-river hydropower schemes in the River Tay catchment (Chapter 3), the longest and one of the most important Scottish salmon rivers, especially noted for its spring-running salmon. **This project aims to assess the potential implications of high-head run-of-river hydropower schemes on fish populations** and identify key areas, such as spawning and nursery habitats, that must be protected. The schemes studied are primarily located within the Tay catchment with one located in the Esk and another in the Shiel catchment.

Historical databases of fisheries data collected for existing run-of-river hydropower schemes were reviewed; those available were obtained from Scottish Environmental Protection Agency (SEPA), Morgan Fisheries Consultancy (MFC), Lochaber Fisheries Trust (LFT) and Hull International Fisheries Institute (HIFI) records. In total, data were available for six SEPA-monitored, eight non-SEPA-monitored and 25 HIFI-monitored run-of-river hydropower schemes. The suitability of schemes for study was determined against set criteria; these included;

- Whether the scheme was currently under construction/operation,
- Whether there were adequate pre-construction/operational data,
- The number of years for which data were available (to account for inter-annual variability),
- The number of control and impact sites (to account for spatial variability),
- The number of years elapsed between the first fisheries survey and the scheme becoming operational, and
- The number of fish/species the river supports; preferably juvenile salmon and trout.

While many schemes were eliminated during the selection process due to their unsuitability for this study, according to the criteria listed above, ten suitable schemes were identified. While at least 2 years' pre construction/operational data were required for the inclusion within this study, with appropriate control and impact sites, a selection of other schemes with limited (<2 years) data and sites were also included to highlight the effects of temporal and spatial variation. These ten schemes were used to address fundamental scientific questions that address the current gaps in scientific knowledge for run-of-river hydropower schemes; and furthermore were used to formulate appropriate science based management protocols for regulation of hydropower.

Specific Objectives

A. Undertake a comprehensive literature review of the potential impacts of run-of-river hydropower schemes on environmental characteristics and biota, with a focus on fish.
B. Assess the status of the fish populations in the study rivers using various fisheries analysis techniques.

C. Quantify the fish population responses to altered flow regimes in depleted reaches in high-head schemes.
D. Isolate the ecological and environmental impacts of run-of-river hydropower schemes from natural variability, especially in relation to fish populations.
E. Identify mitigation options and strategies that could be taken if schemes were to go ahead in certain areas.
F. Undertake impact studies on selected rivers/schemes and provide robust input to develop a future protocol for monitoring such impacts.

Chapter 2 provides a comprehensive literature review to summarise all applicable peer-reviewed and non-peer-reviewed research regarding potential effects of run-of-river hydropower schemes. The diverse array of interacting natural physical, chemical and biological factors that are altered through construction/operation of run-of-river hydropower schemes are discussed.

Chapter 3 outlines the materials and methodology involved during the study, including location of study sites, data collection and data analysis.

Chapter 4 presents results from five run-of-river hydropower schemes that are considered to have adequate pre and post commissioning data to allow sufficient analysis and conclusions to be drawn.

Chapter 5 presents results from five run-of-river hydropower schemes that although are already commissioned, are considered to have weaker data sets in comparison to those in Chapter 4; limitations in the context of data analysis and impact detection are highlighted.

Chapter 6 discusses the importance of environmental monitoring, using Before After Control Impact (BACI) analysis a full impact assessment is conducted and a framework for future monitoring programmes is provided.

Chapter 7 is a general overview and conclusion regarding the implications of run-of-river hydropower scheme upon fish populations; furthermore it outlines future recommendations that are considered of great importance in the aim to improve our knowledge regarding the interactions between man-induced river alterations and fish populations.

CHAPTER TWO

2. LITERATURE REVIEW OF THE POTENTIAL IMPLICATIONS OF RUN-OF-RIVER HYDROPOWER SCHEMES ON FISH POPULATIONS

2.1 Introduction

Over the past decade, concerns about climate change have increased dramatically and shifts towards renewable energy sources have become a focus of attention to reduce carbon emissions. Hydropower schemes are now in place around the world to generate electricity as they are considered a ‘clean (carbon neutral)’ environmentally friendly source of power. These can only be considered “green” energy if environmental implications, especially fish protection and ecosystem services, are taken into account (BHA, 2008). In this context, considerable research has been carried out on large impoundment schemes (e.g. Jager & Smith, 2008), but little attention has been paid to the smaller, run-of-river schemes. This is particularly pertinent given the recent proliferation of small-scale hydropower schemes, particularly of the run-of-river type; they are considered favourable over traditional larger schemes because they are cheaper to construct and maintain. Furthermore, whilst small-scale run-of-river schemes are often presented as an environmentally benign renewable energy source, this description has been challenged by environmental interests who consider small-scale schemes to be damaging to fish and fisheries. This is particularly relevant to these species that rely on regular migrations on a seasonal or life cycle basis (Lucas & Baras, 2001), such as salmonids. Unfortunately there appears to be an acute lack of hard evidence about the impacts of run-of-river schemes on fish and other biota and most information has been gleaned from the better studied impacts of larger hydropower schemes based on substantial impoundments. The potential impacts of large hydropower schemes include:

- changes in fish population structure;
- habitat alterations;
- loss of crucial spawning and nursery habitat;
- loss of biological diversity;
- modifications to water quality and hydrological regimes;
- barriers to fish migration;
- and disruption of longitudinal connectivity threatening fish populations.

The scale and nature of the impacts, and the potential for mitigation vary between large hydropower schemes and run-of-river schemes. These disturbances may potentially lead to changes in fish population structure and thus potential downgrading of the ecological status of the affected water body. This contravenes obligations under the EU Water Framework Directive (WFD) to achieve good ecological status or potential in all surface water and ground water in all Member States (Schmutz *et al.,* 2007; Kataria, 2009).

The Water Framework Directive is a piece of European Legislation that came into force in December 2000 and was transposed into UK law in December 2003. The purpose of the Directive was to establish a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater which:
(a) prevents further deterioration and protects and enhances the status of aquatic ecosystems and, with regard to their water needs, terrestrial ecosystems and wetlands directly depending on the aquatic ecosystems;
(b) promotes sustainable water use based on a long-term protection of available water resources;
(c) aims at enhanced protection and improvement of the aquatic environment *inter alia* through specific measures for the progressive reduction of discharges, emissions and losses of priority substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances;
(d) ensures the progressive reduction of pollution of groundwater and prevents its further pollution, and
(e) contributes to mitigating the effects of floods and droughts

It should be recognised that many of the locations for small-scale hydropower schemes potentially fall under the category of "heavily modified water bodies" because impounding structures that are utilised for modern hydropower development have been in existence for many years on the main stem of many rivers. Nevertheless there remains the obligation to target "good ecological potential" under the WFD and new and existing hydropower schemes may compromise this objective in relation to fisheries.

This chapter reviews the available literature and evidence of the impacts of run-of-river hydropower schemes on fisheries and specifically focuses on small-scale, high-head schemes, although efforts have been made to also identify issues with low-head schemes. As indicated, there is a paucity of information on the impacts of small-scale schemes on fish and other biota, therefore the review draws on and contrasts information from large-scale hydropower schemes about which considerable research has been conducted. The lack of robust information on small-scale schemes highlights the lack of investment in research on small-scale schemes and possibly reflects the marginal economic viability of such schemes and hence inability to contribute towards such fundamental research. Recommendations have been made on the types of mitigation measures that are most effective for each type of hydropower scheme.

2.1.1 Basic design of run-of-river hydropower schemes

Hydropower is considered one of the primary sources of renewable energy and could potentially contribute to meeting the UK's targets for renewable energy. A hydropower scheme of 100 kW installed capacity, for example, will meet the average electricity demand of about 60 houses (assuming other fuel is used for heating) and reduce CO_2 emissions by approximately 200 tonnes per annum. The 2009 Renewable Energy Directive set a target for the UK to achieve 15% of its energy consumption from renewable sources by 2020; Scotland set a more ambitious target with the equivalent of 50% of Scotland's electricity demand to be met by renewable sources by 2015. The basic principle of hydropower schemes is to use the gravitational movement of water to produce electricity. Essentially, this is achieved by passing water over turbines to convert kinetic or potential energy into electrical energy (see Section 2.3.4). The largest hydropower schemes invariably use a dam to store a reservoir of water for electricity generation. By contrast, run-of-river schemes divert a proportion of the river flow through turbines and return the water further downstream. A variety of turbines can be used in run-of-river hydropower schemes with some more damaging than others; this topic is discussed later (Section 2.3.4). In Scotland, data suggest that a total of 102 (68%) hydro developments are storage schemes, with a potential installed capacity of 2148.42 MW and run-of-river schemes accounting for the remaining 49 (32%), with an approximate installed capacity of 62.35 MW (Forrest *et al*., 2008).

Hydropower generation is generally classed into either small-scale (<10 MW installed capacity) or large-scale schemes (>10 MW installed capacity). The hydropower industry further separates small-scale schemes into mini (<1 MW installed capacity), micro (<100 kW installed capacity) and pico (<5 kW installed capacity) schemes. In large schemes requiring the creation of a dam and a reservoir, a pre-determined volume of water produces a more reliable power supply (Jansson, 2002) than small-scale schemes without an impoundment that are dependent on natural river flows. The larger schemes are also able to provide a much more stable power output as the release of water can be controlled to match demand (Hogan, 2005). This is known as hydro-peaking, and causes rapid, large and frequent fluctuations in water flow downstream of hydropower outfalls (Floodmark *et al.,* 2004) (Figure 2.1). Additionally, regulated flows can cause immediate loss of habitat (Brasher, 2003) due to decreased velocity and depth. Such schemes are well documented to have numerous ecological and environmental impacts, affecting the quantity and quality of river habitat (Angilletta *et al.,* 2008). Dams can interfere with the transport of sediment and nutrients along a water course, reduce or alter natural fluctuations in discharge levels, create temperature fluctuations, prevent inundation of floodplains and create wider or shallower rivers (de Leaniz, 2008). Much of the research on the impacts of hydropower development on river ecosystems has been carried out on these large schemes, and will be drawn on where appropriate to elucidate the impacts of small-scale, run-of-river hydropower schemes, the focus of this review. It should be noted, that hydropower schemes, regardless of whether they are run-of-river or storage schemes, are characterised by the same structural elements (impounding structure [potential barrier to migration], water diversion, water intake and outfall) and thus may elicit the same impacts; it is the scale of the operation and thus potential impacts that vary.

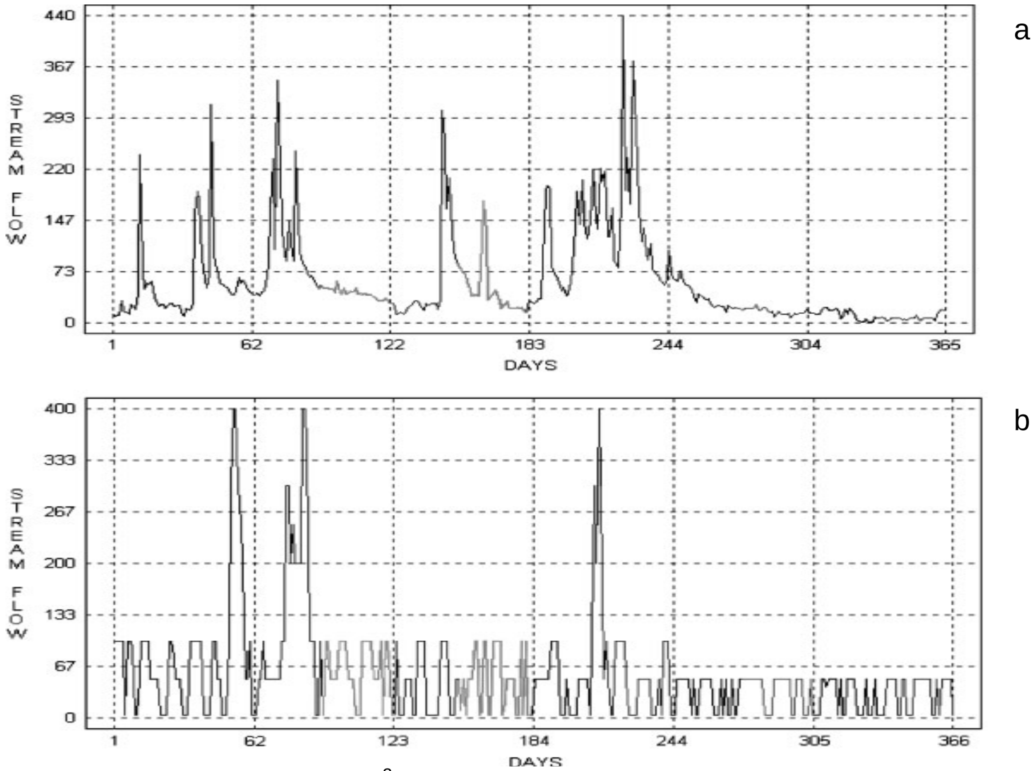

Figure 2.1 Daily discharge (m^3/s) for an average year (1982/83) before (a) and after (b) regulation. In this case the natural regime is replaced by an artificial sequence of flows varying between 4 m^3/s (instream flow) and 50 or 100 m^3/s corresponding to one or two turbines operating, except in situations of maximum release (Cortes *et al*., 2002).

The majority of new schemes in the UK are run-of-river, which have no significant storage of water (considered less than 24 hours). These small-scale hydropower schemes are considered the backbone of electricity production in many EU countries (Bruno *et al.*, 2008). Small-scale run-of-river hydropower schemes can be split into two main types: high-head and low-head. Essentially the main difference is the head of water available for power generation. The head of water relates to the difference in altitude between the intake at the top of the scheme and the floor level of the turbine at the bottom of the scheme, i.e. the outfall (McKenzie, 2007). Low-head hydropower schemes generally have a head of water of < 5 m, while high-head schemes generally have a head of > 50 m. Generally it is considered that the higher the head available, the greater the power output (Hogan, 2005), but in reality power output is a function of discharge volume against head height, and this is particularly relevant to low-head schemes. Low-head schemes are generally constructed on lowland reaches of rivers, with gentle gradients, with high-head schemes based on upper reaches of rivers with steep gradients (Egre & Milewski, 2002). Low-head schemes generally have a greater take of the available flow than high head schemes.

Run-of-river schemes can only operate when there is sufficient flow in the river. However, the amount of water available for power generation is not only determined by the discharge of the river (Larinier, 2008), but is dependent upon the abstraction regime of individual schemes, which relates to the discharge. Consequently, run-of-river schemes are thought to cause lower disturbance and impact to stream ecology (Batrich *et al.,* 2004). Additionally, impacts are considered to be less damaging as they do not lead to fragmentation of riverine habitat, with the condition that in-stream flows remain sufficient (Habit *et al.,* 2007). However, stretches between the intake and outflow can experience severely reduced flows, which can cause changes to habitat characteristics, water quality and migration movements (see Section 2.2.4). Nevertheless, run-of-river schemes require an impounding structure, and many are re-developments of existing sites, such as old mills. Generally run-of-river schemes are located on existing structures or require construction of small weirs, the impacts of which on fish communities are poorly understood (Benstead *et al.,* 1999; Kingsford, 2000; Aarestrup & Koed, 2003; Poulet, 2007; de Leaniz, 2008).

2.1.2 High-head run-of-river hydropower schemes

High-head run-of-river hydropower schemes are usually located on steep, fast-flowing streams and rivers (Figure 2.2). In essence, high-head run-of-river hydropower schemes divert a proportion of the river flow to drive a turbine before the water is returned to the watercourse at a downstream location without storage (Figure 2.3). A weir is usually built to create a head of water for abstraction, with the intake guarded with a screen to prevent entrainment of fish (Figure 2.3). Abstracted water is channelled down a

Figure 2.2 A steep, fast-flowing stream considered suitable for a high-head run-of-river hydropower scheme.

pipeline (often buried for aesthetic reasons) to the turbines in a powerhouse before release at the outfall position.

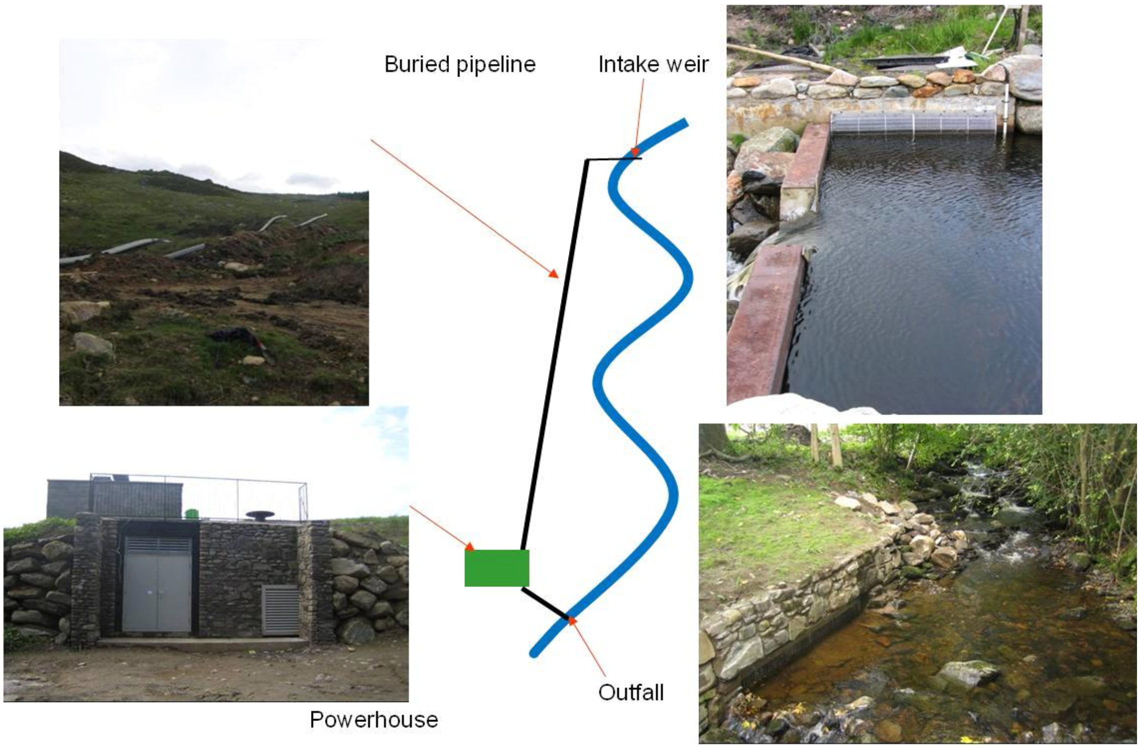

Figure 2.3 Typical layout of a high-head run-of-river hydropower scheme.

The outfall point is guarded with a screen and the discharged flow is usually dissipated to ensure fish are not attracted to the outfall (Figure 2.3). Without storage, these schemes operate according to the flow that is present at a given time. Streams suitable for hydropower generation often have highly dynamic flow regimes (Figure 2.4) and are very responsive to rainfall inputs. Typically, variations in flow encompass low flows during summer, interspersed with spates, followed by more frequent and larger spates in the autumn and winter (Figure 2.4). The volume of water abstracted is variable and dependent on the economic and environmental requirements of the scheme. The greatest abstraction usually occurs at high to moderate flows, with the level of abstraction tailing off as lower flows are approached.

2.1.3 Low-head run-of-river hydropower schemes

Low-head run-of-river hydropower schemes are often located on existing in-channel structures, such as weirs and mills, where the fall of water is often < 5 m (Figure 2.5). These include not only old mill sites but also those built for navigation, flood defence, abstraction or aesthetics. For most of these, any impact on the river environment when they were constructed was not understood or even considered. Therefore many low-head schemes are located on larger rivers and operational success relies on sufficient flow over the weir to allow diversion for power generation.

Water is often diverted via an existing mill leat (Figures 2.5 & 2.6) and channelled through a screened turbine before release downstream. In these cases the turbine can be some distance downstream from where existing mill structures are located, resulting in long reaches of depleted flow (Figure 2.6). Alternatively the turbine can be on or close to the weir structure, with water returned to an existing weir pool (Figures 2.6 & 2.7) which avoids depleting any length of watercourse of its flow. In Scotland, low-head schemes are usually located on lades and generally involve some diversion of water. Abstraction is variable and dependent on the flow regime of individual rivers. The head of water available is influenced not only by the height of the weir (and water flowing over) but also the river level downstream of the existing impoundment (Jager & Bevelhimer, 2007).

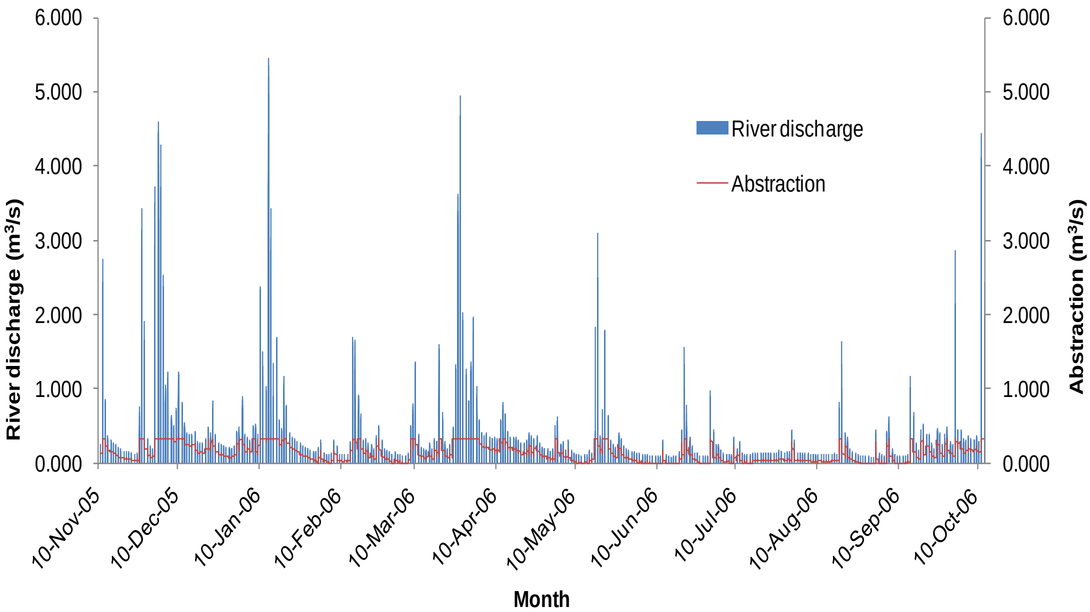

Figure 2.4 Normal and modelled abstracted flow regime (m^3/s) in Rottal Burn between November 2005 and November 2006.

Figure 2.5 Existing weir on River Ribble, Settle (left), originally used to divert water to a mill via a mill leat (right), now used for diversion to a hydropower turbine.

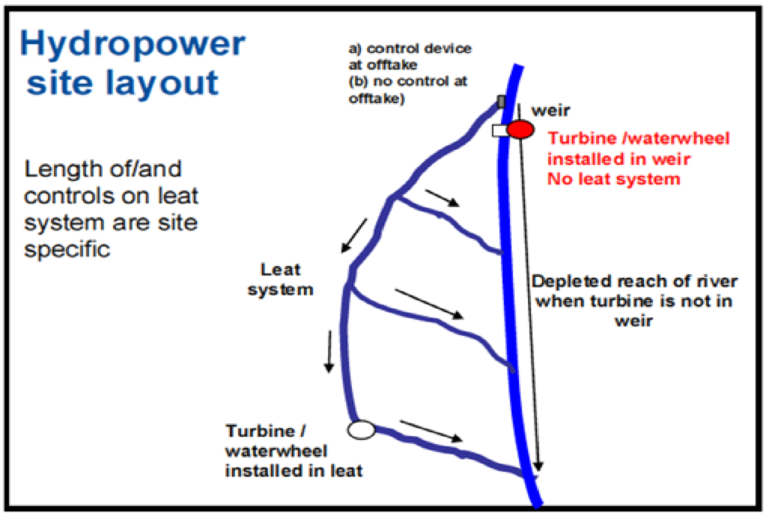

Figure 2.6 Typical low-head hydropower layout on a mill leat. This leat system has overflows (represented by arrows) to control the flow of water in the system. (EA, 2009a).

Figure 2.7 Archimedean hydropower turbine on a weir structure (photo courtesy J. Aldrick).

In the majority of cases, low-head schemes operate on a continuous basis, when sufficient water is available, but occasionally water is stored behind the impounding structure and the schemes operate mainly when there is peak demand for electricity. Because the head is low, compared with high-head schemes, the volume of water used per unit of power is high. Such schemes are generally designed to use the long-term daily mean flow of the river when on full load. Because low-head schemes have little provision for storing water, the economic imperative is to use as much as possible of the total river flow at any time, although the recent introduction of more modern and thus efficient turbines with higher power-generating capacities has somewhat offset this requirement. Modern turbines can operate efficiently with flows lower than one quarter of their full load design flow (Irish Fisheries Board, 2007). Accordingly, turbines can continue running during dry weather flows.

2.2 Potential impacts and issues of run-of-river hydropower schemes

2.2.1 Introduction

Small-scale hydropower plants are generally of the run-of-river type, and may exhibit differences in design, appearance and impact from conventional large hydroelectric schemes (Anagnostopoulos & Papantonis, 2007). In general, however, many of the characteristics of large impoundment hydropower schemes and run-of-river schemes are similar (Table 1.1) and it is more a question of scale in terms of potential impacts. Consequently it is possible to elucidate potential impacts of run-of-river schemes from the much better studied large impoundment schemes. Nevertheless, run-of-river hydropower schemes are considered to be more environmentally friendly than traditional hydropower schemes (A. Butterworth, pers. comm.) as the social and environmental impacts are suggested to be much less severe in the former. However, there remains a paucity and clarity of information on the ecological effects and responses of fish communities to such run-of-river hydropower schemes (Habit *et al.,* 2007) to substantiate these views (Copeman, 1997) as they are poorly understood and have not been adequately studied (March *et al.,* 2003). To acknowledge this lack of empirical support, the Scottish Government recently released a policy statement to ensure that whilst they acknowledge the valuable contribution that hydropower generation makes to meet the renewable energy targets, they are still undertaking appropriate measures to ensure Scotland's water environment is adequately protected from significant adverse impacts (SEPA, 2011). Furthermore, after an initial rush for small-scale hydropower schemes in North America in the 1980s and 1990s, these schemes have largely been abandoned because of the potential impacts they have had on the environment and their failure to deliver economically viable power production (M. Kondolf, pers. comm.).

Run-of-river schemes have been thought to essentially maintain a natural flow regime (Jager & Bevelhimer, 2007; Enders *et al.,* 2009), and consequently allow for a biologically 'friendly' flow scenario, especially with regard to migratory species such as Atlantic salmon. However, this is not accurate; although they usually do not impound the system, they modify flow regimes, and abstraction by run-of-river schemes can result in a depleted reach between the intake and outfall. The term "depleted reach" indicates the stretch of river between the intake and outfall that experiences reduced flow (specific discharge amounts unknown) due to abstraction and is not implying a completely dry river. There are also a number of other potential impacts of small-scale run-of-river hydropower schemes on the ecology of the affected river reach (Figure 2.8). Therefore, a common misconception is that hydropower is harmless to the environment. The inherent variability and poor understanding of how far river flows can be altered beyond the natural range before unacceptable ecological change becomes apparent (Gladwell, 2000; Merrett, 2007), is allowing hydropower to become a worldwide political issue, the impacts of which are open to debate. The question remains, however, about how much flow can be diverted from a river before the ecology is damaged (Hogan, 2005). This is considered the primary impact of run-of-river schemes, as the hydraulic regime over the abstracted reach is altered (Copeman, 1997), experiencing lower flows than normal. The aim of this section is to review the potential impacts and issues of run-of-river hydropower schemes on fisheries based on available peer-reviewed and grey literature and use this information to identify mitigation measures to ameliorate or eliminate the impacts. To achieve this, the various issues highlighted in Figure 2.8 will be examined in detail in the following sections.

2.2.2 Impoundment structures

Rivers throughout the world have been modified with locks, dams or weirs to optimise water levels for navigation, generate electricity or for agricultural land use (Knaepkens *et al.,* 2005). The creation of these man-made structures has resulted in the fragmentation of freshwater ecosystems, with an associated isolation of populations (Winston *et al.,* 1991; Labonne & Gaudin, 2005; Schilt, 2007). This disruption to longitudinal connectivity can compromise the ecological integrity of river systems (Jungwirth, 1996), disrupting the life cycles of many species, and the maintenance of healthy ecosystems (Odeh, 1999a).

The potential increase in fragmentation of riverine ecosystems by the proliferation of small, low-head barriers (Petts, 1990) is likely to further impact migration processes and population structures (Labonne & Gaudin, 2005), but this remains to be quantified in long-term studies despite many rivers having been impacted for centuries (Benstead *et al.,* 1999). An indication of the potential impacts of such barriers on fish and fisheries can be gleaned from larger structures, although it should be recognised that structures for run-of-river hydropower schemes tend to be considerably smaller and are often passable under certain or sometimes all flow regimes. Impounding structures pose threats to the maintenance of healthy ecosystems by disrupting sediment dynamics, altering riverine biodiversity, composition and abundance (Renofalt *et al.,* 2010), reducing access to feeding, spawning and nursery habitats (Odeh, 1999a; Trussart *et al.,* 2002; Gosset *et al.,* 2006; Mader & Maier, 2008), and blocking or delaying the movements of migratory fish (Lucas *et al.,* 2009) such as Atlantic salmon, sea trout and eels, many of which are adapted to the rivers' continuity (Stanford *et al.,* 1996; Corbacho & Sanchez, 2001; Morita & Yamamoto, 2002; Fette *et al.,* 2007; Godinho & Kynard, 2009).

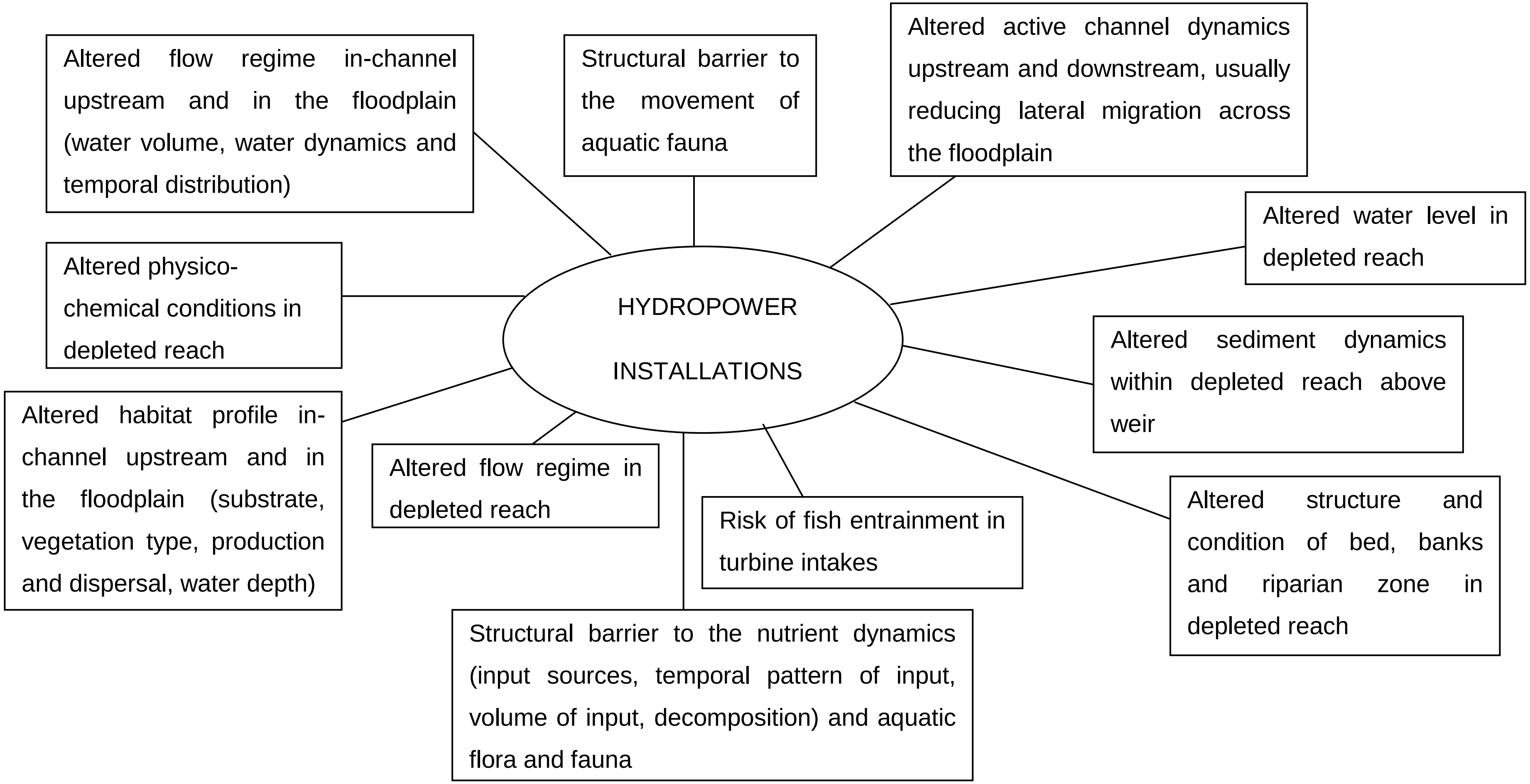

Figure 2.8 Range of possible alterations typically associated with hydropower dams with subsequent biological alterations (modified from Vovk-Korz *et al.,* 2008).

2.2.3 Longitudinal connectivity

The requirement of diadromous species, such as salmonids, eels, lampreys and shads, for free passage is well known (Northcote, 1984; Nunn *et al*., 2008; Lucas *et al.,* 2009), but there is also growing evidence that many more species, such as cyprinids, percids and pike (*Esox lucius* L.), exhibit migratory behaviour (Harris & Mallen-Cooper, 1994; Lucas *et al*., 1998; Bolland *et al*., 2008; Nunn *et al*., 2010), which can equally be disrupted by impoundment structures. The decline in some cyprinid populations, especially of obligatory rheophilic species, in many river systems has been linked to interruption of cyclical movements by dams and weirs (Lucas & Frear, 1997). Maintenance of longitudinal (and lateral) connectivity is therefore essential to the viability of fish communities (Bunn & Arthington, 2002) as it enables continuity of the life cycle through access to their necessary habitats (Aarts *et al.,* 2004; Hirzinger *et al.,* 2004; Koel, 2004), and dispersal of riverine organisms, which ultimately contributes to organism genetic fitness (Hartvich *et al.,* 2008).

Fragmentation of rivers by dams and weirs potentially disrupts the life cycles of many fish species by interfering with both upstream and downstream movement between key habitats, such as those used for spawning, foraging and refuge (Northcote, 1995; Jungwirth, 1998; Jungwirth *et al.,* 1998; Lucas & Baras, 2001). Indeed, fragmentation has been implicated in declines in riverine fish populations and their habitats (Jansson, 2002; Trussart *et al.,* 2002), changes to ecological processes and communities, and recruitment bottlenecks in impacted systems (Ward & Stanford, 1995; Jansson *et al.,* 2000), thus presenting a major concern for fishes (Ormerod, 2003). Ultimately, isolation (Bunn & Arthington, 2002) can lead to a decline, or even extirpation, of populations (Mallen-Cooper, 1999, 2000; Larinier, 2000; Penczak & Kruk, 2000; Bunn & Arthington, 2002; Gehrke *et al.,* 2002; Meldgaard *et al.,* 2003; Masters *et al.,* 2006; Lucas *et al.,* 2009) resulting from incomplete or unsuccessful spawning migrations that limit reproductive potential (Jungwirth, 1996). Although this evidence is based upon dams and weirs, the principles remain the same for potential impacts of small-scale schemes as impoundments can be caused by these features in run-of-river schemes; ultimately the presence of weirs or barriers used in run-of-river schemes provides potential for fragmentation, undoubtedly bringing knock-on effects such as interference of longitudinal movement. In the northern hemisphere in particular, dams are suggested as the most obvious and widespread anthropogenic alteration to river ecosystems (Dynesius & Nilsson, 1994; Nilsson *et al.,* 2005); water abstraction and pollution are additionally considered key stressors affecting the integrity of running waters (Pielou, 1998; Heinz Center, 2002; Millennium Ecosystem Assessment, 2005).

Weirs and dams can also increase the vulnerability of migratory fish to anglers, potentially leading to overfishing, alter hydrologic and geomorphic regimes (Gehrke *et al.,* 1995; Ligon *et al.,* 1995; Galat & Lipkin, 2000; Lytle & Poff, 2004) and thus block migration routes and interfere with navigational cues (Drinkwater & Frank, 1994). Furthermore, weirs and dams can cause declines in biodiversity and alter food-web structure (Power *et al.,* 1996, Wootton *et al.,* 1996, Pringle, 1997), and exacerbate the effects of opportunistic predators (de Leaniz, 2008), in some cases leading to the loss of entire populations (Meldgaard *et al.,* 2003).

Low-head types of barriers are abundant in the UK (Lucas *et al.,* 2009; see Figure 1.1, for potential locations), with some 16,000 in England and Wales alone (Environment Agency, 2009a). However, their impacts on catchment fragmentation are often overlooked as low-head structures are often not regarded as barriers to fish movements, thus their effects are poorly understood (Aarestrup & Koed, 2003; de Leaniz, 2008). Nevertheless, such barriers may impede eels from colonising large parts of catchments, thus reducing upstream density and the production of adults (A. Butterworth, pers. comm.). Furthermore, radio-tagging studies have shown that fish are reluctant to move downstream over or through regulating structures, and often return upstream when confronted by a weir (Haro *et al.,* 2000; Behrmann-Godel & Eckmann, 2003). Similarly, Lucas & Frear (1997) reported that out of 23 barbel (*Barbus barbus* (L.)) tracked, only 15 attempted to pass a flow-gauging weir, with only six of these being successful. Those that passed the weir moved considerable distances upstream, whereas unsuccessful individuals moved downstream, highlighting the impact of even small obstructions. Ovidio & Philippart (2002) also found that fish unsuccessfully negotiating a barrier returned downstream to a distance of several hundred of metres. These fish waited for environmental conditions to improve, such as increased water level and temperature (Bjornn & Peery, 1992; Trepanier *et al.,* 1996) before moving upstream. Similarly, a radio-tracking study undertaken on the Gave de Pau, France, found only 16 out of about 30 obstructions (of which 20 were small-scale hydro-schemes) allowed all migrating fish to pass without significant delays, 10 structures were more serious obstacles to migration in terms of delays or blocking part of the population, but were still negotiable, and five structures, of which several were located on the downstream part of the migration route and of older design, were major obstacles (Larinier, 2008). Thorstad *et al.* (2003), investigating the upstream migration of Atlantic salmon at a power station on the River Nidelva, southern Norway, found that not one of 10 salmon that passed the outlet and reached the residual flow stretch passed the dam.

These findings are of importance because numerous studies have revealed the precise nature of salmonids when choosing spawning grounds (Gore & Hamilton, 1996; Gosset *et al.,* 2006). Salmon and brown trout both tend to return to their natal river, even their natal tributary, to spawn (Crisp, 2000), although the exact spawning location may differ with age. Youngson & Hay (1996) and Okland *et al.* (2001), for example, found that multi-sea-winter fish spawned higher upstream than grilse. This is potentially problematic if the presence of barriers, such as weirs, causes age classes (genetically distinct life history types in the case of Atlantic salmon) to interbreed if preferred spawning grounds are unreachable. Using radio tagging, Lucas *et al.* (2009) found that 64% of adult river lamprey (*Lampetra fluviatilis* (L.)) passed one weir on the River Derwent and 17% passed a second weir; but showed that high flows were the crucial factor affecting their ability to bypass the weirs. In addition, even slight modifications to local habitats can influence the suitability for certain species and thus can lead to changes in fish species assemblages. Any alterations to habitats potentially create conditions to which native biota may be poorly adapted (Poff *et al.,* 1997). Many species such as river lamprey, brook lamprey (*Lampetra planeri* L.) and sea lamprey (*Petromyzon marinus* L.) and salmon are sensitive to habitat alteration (Lucas *et al.,* 2009).

Although it is widely assumed that the height of a barrier is positively correlated to the difficulty of fish passage, this is not necessarily the case because fish passage is dependent upon a range of factors, including hydraulic characteristics, river flow, water temperature, and the species and size of fish (Larinier, 2001); low structures can thus be as difficult to pass as much higher barriers.

The engineering work associated with the construction of small dams and weirs can also greatly affect river ecosystems (e.g. physical removal of habitat), potentially resulting in a loss of fish-spawning grounds. Furthermore, increased sedimentation, as a result of reduced flows downstream and 'ponding' upstream, and disturbance during the installation of schemes, can clog the interstitial spaces in gravel beds, which impedes the flow of highly-oxygenated water through the redds, potentially reducing productive capacity (Crisp, 2000) (see Section 2.2). This highlights the importance of high flows during the autumn / winter when salmonids are preparing to spawn. Although hydro developers may regard high flows during winter as an opportunity to take advantage of (extra) water, high flows are essential to remove fine sediments from gravel beds and maximise the incubation success of salmonid eggs and alevins.

In the context of construction of barriers associated with hydropower schemes, it should be recognised that large numbers of weirs have been a feature of our river systems since before the 19th century and as such fish populations have always had to try and negotiate these barriers. Hydropower schemes, however, are commonly installed on these existing weirs and they are becoming increasingly more difficult to negotiate because of the changes in hydraulic conditions around the weir caused by diversion of water flow associated with hydropower generation. In cases where weirs cannot be removed, or are unlikely to be removed, fish passes should be fitted regardless of which species are within the given stretch of river. It is equally important that fish species within the given river are identified so the correct fish pass design can be produced (see Section 2.3.2.2).

To conclude, impounding structures associated with small-scale hydropower schemes may impede the upstream migration of many fish species in the UK. Logically, any loss of access to upstream reaches may cause deterioration in ecological class status and impact on obligations under the WFD. Since promotion of renewable energies, including hydropower, is a recognised strategy to mitigate the impact of climate change, and is the subject of an EU Directive, applications for proposed run-of-river hydropower schemes are noticeably increasing. Under these circumstances, options should be explored to benefit both the ecosystem and human needs.

2.2.4 Potential impacts of depleted flow reaches in rivers

Flows in UK rivers are frequently compromised because of increasing demands for water abstraction, including diversion of flow for run-of-river hydropower schemes (Acreman *et al*., 2008, 2009). One of the key concerns of run-of-river hydropower schemes is the impact on fish and fisheries of flow depletion/dewatering in bypass sections of river and the depleted reach of the main river. Unfortunately there is little information on this issue with respect to small-scale, run-of-river hydropower schemes, so many of the potential impacts have been identified from literature relating to larger impoundments and the issues associated with altered flow regimes. Nevertheless, the UK Technical Advisory Group (TAG), has developed environmental flow standards for rivers based on macrophytes, invertebrates and fish (WFD 48 and 82; Acreman *et al*., 2008, 2009). These standards are discussed in more detail in Section 2.3.3 related to mitigation measures. However, the importance of providing adequate and appropriate flow regimes is critical because European Directives impose an obligation to protect the habitats of threatened species (Habitats Directive) and ensure that all water bodies attain good ecological status or good ecological potential (Water Framework Directive); a key requirement for achieving or maintaining good ecological status (Cowx *et al.*, 2004).

The importance of hydrological variability for shaping the biophysical attributes and functioning of river ecosystems is well recognised (Ward & Stanford, 1979; Poff *et al.,* 1997; Richter *et al.,* 2003; Kennard *et al.,* 2010). Viewed as a “maestro” (Walker *et al.,* 1995) or “master variable” (Power *et al.,* 1995), stream flow is responsible for shaping the fundamental ecological characteristics and physical habitat of all river ecosystems (Almodovar & Nicola, 1999; Bunn & Arthington, 2002, Gilvear *et al.,* 2002), and maintains the natural biological diversity of riverine species, many of which have evolved life cycles and life history strategies that are intrinsically linked to natural flow regimes (Enders *et al.,* 2009). River flow maintains the structure, species, communities, processes and functions that provide ecosystems with specific characters (Acreman & Ferguson, 2010). However, human society has modified and altered flow regimes to provide dependable ecological services such as water supply, flood control, recreation, navigation and hydropower production (Kennard *et al.,* 2010). Effects from hydrological alterations are particularly evident on fish (Nilsson & Brittain, 1996; Welcomme *et al.*, 2006), where changes in aquatic ecosystems can restrict or hinder fish migration, affect water quantity and quality, increase predation and cause direct damage and stress (Schilt, 2007), ultimately changing the landscape structure and leading to an impoverishment of natural diversity (Nilsson, 1996; Richter *et al.,* 1997). Since flow is considered to be the major driver of all ecological processes in rivers, alterations of flow regimes through hydropower development may radically change ecosystems (Walker & Thoms, 1993; Kingsford, 2000), and threaten the biodiversity and ecosystem functions of rivers (Postel & Richter, 2003; Nilsson *et al.,* 2005; Dudgeon *et al.,* 2006; Poff *et al.,* 2007). Regardless of their size, hydropower dams and physical manipulations to rivers are generally associated with alterations in stream flow, which can result in substantial ecological impacts (Anderson *et al.,* 2006).

2.2.4.1 Changes in flow regime caused by hydropower

The critical component in the economic viability of run-of-river hydropower schemes is the volume of water passing through the turbines. There is therefore a requirement for sufficient abstraction or diversion of water for economic viability of the hydropower scheme, which ultimately results in a reach of river between the intake and outfall with depleted flow. Flow duration curves (Figure 2.9) for run-of-river schemes suggest the main loss of flows is in the middle of the flow range. Ideally, abstraction regimes should maintain peak flows and protect low flows, and where possible reflect the natural flow regime, i.e. only a fixed proportion of the intermediate flow is abstracted (Figure 2.9a). However, large abstractions can reduce spate flows (see Figure 2.1); producing flatter flow duration curves (Figure 2.9b). It should be noted that the low flows, i.e. Q95 and lower, in run-of-river schemes are generally not affected because the flows are either protected or inadequate to drive the turbines so the schemes do not operate.

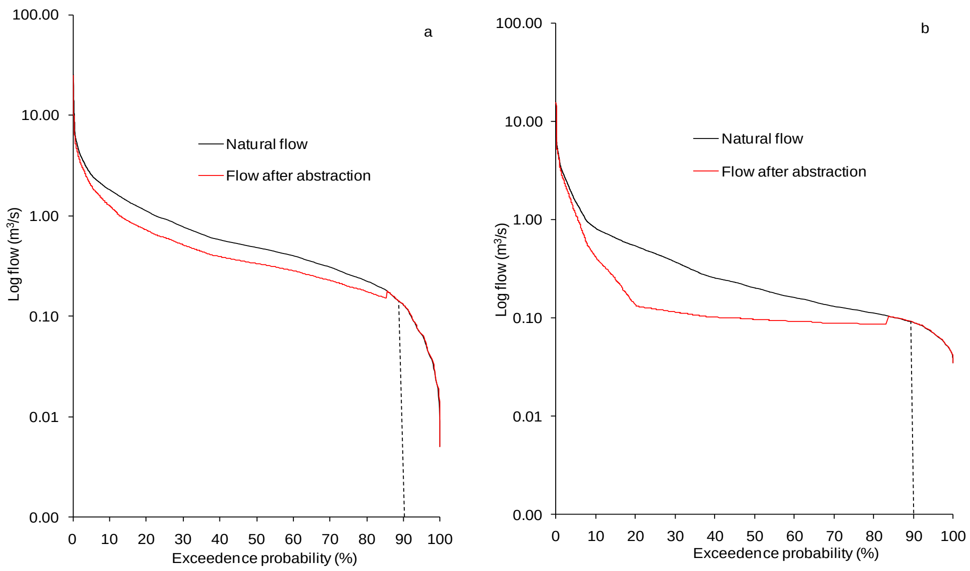

Figure 2.9 Flow duration curves for two contrasting high-head run-of-river hydropower regimes (dotted line indicates hands-off flow at Q_{90}, see Section 2.3.3).

2.2.4.2 Reductions in flow

Reductions in river flow can cause substantial ecological impacts and are frequently associated with large impoundment hydropower schemes (Armstrong *et al.,* 1998; Saltveit *et al.,* 2001). However, run-of-river schemes, which generally have small impoundment weirs, may also cause ecological impacts, as the reaches between water diversion and release have a reduced flow. This may impact on the ecology of the depleted flow reaches (Copestake, 2005), including an altered availability of habitat features (Whiting, 2002), a reduction of fish biomass (Baran *et al.,* 1995), loss of river continuity and impediment to fish movements (A. Butterworth, pers. comm.), risks of fish entrapment at intakes, build up of sediment at outfalls and sudden releases at turbine houses. Kubecka *et al.* (1997), whilst investigating 23 bypass- or mill stream-type small hydropower stations reported considerable loss in wetted area, thus essential habitat, as a result of reduced stream flow during abstraction. In some instances, when no water passed over the diverting weir, the wetted area decreased by 40-50% compared with 0-30% reduction in wetted area when a reduced flow was present.

Water flow is suggested to be the primary factor controlling when salmonids enter rivers (Jonsson & Jonsson, 2011); increases or decreases in river flow appear to be important for the timing of the ascent (Banks, 1969; Jonsson, 1991). It is widely accepted that Atlantic salmon preferentially migrate into rivers during periods of high flow (Ladle, 2002; Old & Acreman, 2006), and it has been suggested that fish respond to hydraulic cues in the near field using the mechano-sensory lateral line system (Giske *et al.,* 1998). Consequently, modifications of flow regimes could result in resident and migratory species being denied key environmental cues (Rosenberg *et al.,* 1997). Flow changes associated with small-scale run-of-river schemes could potentially alter river-wide flows and thus migration cues; disruption of free movement over a barrier (weir) or through the depleted flow reach are likely to be the major problems encountered with maintaining migratory pathways. De gaudemar & Beall (1998) reported over ripening of Atlantic salmon eggs because of a forced delay of spawning of ovulated females, this not only reduces egg viability but also shortens the spawning period and thus increases egg retention.

2.2.4.3 Changes in habitat characteristics

Habitat complexity increases with water depth, water velocity and cover (Gorman & Karr, 1978; Schlosser, 1982; Felley & Felley, 1987), resulting in increases in the richness of aquatic fauna. Reductions in flow like those in the depleted reaches, can therefore reducing the amount of potential spawning and/or nursery habitat and ultimately affect the number of species able to utilise the area. This topic is covered in more detail in section 4.7

2.2.4.4 Changes to species assemblage

Physical conditions in depleted reaches may also favour tolerant fish species with opportunistic life histories over those whose reproductive requirements are more specific and complex (Anderson *et al.,* 2008), thus altering species compositions. Indeed, Magoulick (2000), Lake (2003) and Matthews & Marsh-Matthews (2003) reported reductions in native and intolerant species related to reduced flows during droughts, and Pusey *et al.* (1993) made similar observations; where tributaries had experienced extended periods of low flow, with small, physiologically-tolerant and generalist species becoming dominant. Kubecka *et al.* (1997) reported that water abstraction in low-head hydropower schemes in the Czech Republic resulted in low flows and thus depleted reaches, causing changes in species composition from large-bodied (adult brown trout, chub (*Leuciscus cephalus* (L.)), dace (*Leuciscus leuciscus* (L.)) and grayling (*Thymallus thymallus* (L.)) to small-bodied fish (juvenile trout, minnow (*Phoxinus phoxinus* (L.)), bullhead (*Cottus gobio* L.), stone loach (*Barbatula barbatula* (L.)) and gudgeon (*Gobio gobio* (L.)). Additionally, diverting weirs imposed migration barriers for resident fish in 30% of small hydropower stations (Kubecka *et al.,* 1997). It is now widely accepted that a naturally variable flow regime, rather than just a minimum low flow, is required to sustain freshwater ecosystems (Poff *et al.,* 1997; Bunn &

Arthington, 2002; Postel & Richter, 2003; Annear *et al.,* 2004; Biggs *et al.,* 2005, Poff & Zimmerman, 2010).

2.2.4.5 Impacts on sediment movement

As run-of-river schemes result in depleted reaches between the intake and the outfall, there is also concern regarding the effects of increased sedimentation – a term often used in the context of fine sediment impacts on salmonids (Sear *et al.,* 2008). Excess sedimentation can cause numerous effects including gill irritation, impede movement, alter foraging behaviour, alter blood physiology and sometimes induce mortality (Bash *et al.,* 2001; Kemp *et al.,* 2011). However, the biggest concerns are the changes sedimentation has on salmonid spawning and rearing habitat and ultimately fish reproduction. Salmonids prefer spawning grounds with clean gravel and highly-oxygenated, fast-flowing water (Crisp, 2000). Hvidsten (1985) suggested that reductions of 0+ trout in the River Nidelva were due to stranding and consequently increased mortality caused by frequent changes in flow and thus water level. Females select their spawning sites according to water depth and velocity, substrate composition (e.g. grain size, compaction/stability, porosity), the occurrence of up- or down-welling flows, and availability of nearby cover (Heggeberget, 1988; Verspoor *et al.,* 2007). Redds are created in cold, fast-flowing waters, not only because of the high dissolved oxygen concentrations but also to mix the sperm and eggs and ensure efficient fertilisation. The eggs are buried under 10-15 cm of gravel, thus a habitat comprised of clean gravel is of great importance. High water velocities ensure the gravels contain low concentrations of fine sediments such as sand and silt. This highlights the importance of high flows, especially in the winter during the spawning and incubation period of salmonids, and suggests that maintaining naturally high flows by a limited abstraction rate during the spawning season will minimise the chance of spawning habitat being smothered by fine sediments.

Another potential issue is accumulation of sediment upstream of a weir. The intake can, however, be designed appropriately so that sediment can be flushed downstream during periods of high flow. Furthermore sediment can be transported and reintroduced to the river, but this should only occur during periods of high flow, at locations that will not inhibit the movement of migratory fish species and outside periods when fish are likely to be spawning or the eggs incubating in the gravels, i.e. pre-emergence.

2.2.4.6 Migration movements

Low flows can also change the behaviour of migratory species (see Section 2.3). It has been suggested that fish movements may occur in direct response to changes in flow

(Vehanen *et al.,* 2000; Murchie *et al.,* 2008), with the upstream migration of adult salmon related to increased discharge and fish moving on the receding phase of the spate (Alabaster, 1990). This is supported by the work of Saunders (1960) who observed Atlantic salmon entering a river during autumn freshets but during periods of low flow they remained at the head of the estuary. Potter (1988) reinforced this reporting Atlantic salmon entering the River Fowey, England, during periods of increasing freshwater discharge and further identified that low river flow delayed their migration into the river. Furthermore, Arnekleiv & Kraabol (1996) reported that ferox trout did not pass an outlet channel from a power station when the residual flow decreased below 20 m^3/s and an artificial freshet of 60 m^3/s could initiate their upstream migration, thus increased water level can indeed facilitate upstream migration.

Although it is now generally accepted that high flows are preferred by migrating fish (Weaver, 1963; Banks, 1969; Ladle, 2002), some flows can be too high for salmon entry (Old & Acreman, 2006). It is now understood that, in some cases, salmon may enter a river but return to the estuary and move to other rivers when the influence of flow is constrained by flow modification (Old & Acreman, 2006), but this response is related to larger schemes and is unlikely to occur in smaller schemes. However, in relation to small-scale run-of-river schemes, Arnekleiv & Kraabol (1996) found that upstream migration of brown trout stopped when a depleted river reach had a flow that was 10% of the turbine flow. When the river flow was greater than 30% of the turbine flow, 50-60% of brown trout moved through the depleted reach, but the remainder stayed in the vicinity of the outflow or returned downstream. This is welcoming as the use of the Q_{mean} and hands-of-flow approach as mentioned in the EA Good Practice Guidelines (GPG) ensures that flow within the depleted reach of the river is generally greater than 30% of the turbine flow.

Although low flows can limit adult salmon entry into rivers (Gibbins *et al.,* 2008) and interfere with movement upstream to spawning habitat, as well as reducing habitat area and quality (Crisp, 2000), fish may prefer decreasing flows after river entry (Gowans *et al.,* 1999; Lilja & Romakkaniemi, 2003). Jonsson *et al.* (2007) reported that the migratory activity of Atlantic salmon and brown trout decreased in the River Isma at very high flows and Keefer *et al.* (2004) reported that chinook salmon (*Oncorhynchus tshawytscha* (L.)) in the Columbia River moved more slowly when the water discharge was high, thus supporting the fact that high flows can halt upstream migration (Jensen *et al.*, 1989). Furthermore, Trepanier *et al*. (1996) found that the ascent of landlocked Atlantic salmon correlated negatively with water flow, suggesting further that fish preferred low water phases for their ascent. By contrast, Boubee *et al.* (2008) reported an increased activity of eels (*Anguilla* spp.) with rising water levels, with migration towards the sea only occurring when there was sufficient flow, and Jansen *et al.* (2007) reported similar observations, with eels adjusting their migration route in response to the volume of river flow, a behaviour also observed in Pacific salmon smolts (Kemp *et al.,* 2008). Recaptures of carlin-tagged salmon showed that large individuals returned from the feeding areas in high seas to coastal waters in June-July but did not enter the river before September-October (Jonsson *et al.,* 1990a,b, 2007) when flows had increased. Furthermore, Mitchell & Cunjak (2007) found a strong relationship between the total number of adult Atlantic salmon returning within a year and maximum water discharge during the migration period. Thus, reduced water flow during the upstream spawning migration had a significant effect on adult return. Conversely, during periods of low flow on the Mokau River, New Zealand, eels (*Anguilla* sp.) attempting to migrate downstream past a hydropower facility appeared to search for an outlet and either passed through turbines, were delayed upstream or made use of an artificial bypass channel to aid migration (Boubee & Williams, 2006). However, when flows increased and water spilled over a dam, eels showed a preference for this route downstream, although some were entrained or impinged on screens (Boubee & Williams, 2006).

Boubee & Williams (2006) suggested that if flows over obstructions are reduced, fish passage can be reduced or even terminated. Additionally, it has been reported that mortality rates through turbines may be directly linked to river flow rates, with mortality rates being greatest during periods of low flow (Hadderingh & Bakker, 1998; Jansen *et al.,* 2007; see also Section 2.2.5). It has however also been reported that the relationship between water flow and rate of upstream migration differs with the size of fish (Keefer *et al.,* 2009). While Van den Berghe & Gross (1989) reported large coho salmon (*Oncorhynchus kisutch* (L.)) females entering the creek in the Deer Creek Junior at peak discharge, progressively smaller ones were observed entering as water level reduced. Furthermore, river entry requirements have also been suggested to depend on the amount of spawning time remaining. Tetzlaff *et al*. (2005) found that Atlantic salmon movements were triggered by increased water discharge at suboptimal flows in the Girnock Burn, Scotland, and that the threshold flow for migration decreased with decreasing time to spawning. Close to spawning salmon needed less water to ascend rivers than they did earlier.

2.2.4.7 Changes in water quality

Reductions in flow can also influence water temperature, both of which are important abiotic factors that change after the regulation of rivers (Ward & Stanford, 1979; Petts, 1984). Thermal cues influence salmon spawning, the migration of smolts and the emergence of larvae, thus anthropogenic changes in temperature may lead to mis-matches between environmental conditions and life cycles (Stenseth & Mysterud, 2002). Other studies, however, did not find fluctuating river levels to influence movement patterns, habitat use or displacement (Bunt *et al.,* 1999; Robertson *et al.,* 2004).

In summary, depleted flows may result in losses in fish production, reduction in reproductive output and impede upstream migration/movement of adult fish. It should be noted, however, that these are unlikely to be important in high-head schemes where the diversion is around impassable falls or cascading flow regions, assuming the water is returned to the systems immediately downstream of the natural obstruction and that the intake is immediately upstream of the obstruction. This will not be the case however, where local topography and / or land ownership prevents the water being returned immediately downstream of the impassable barrier, or when the outfall is further downstream of a barrier for sufficient head of water for generation. In these cases, it is possible there is an impacted (depleted flow) length of river immediately downstream of the barrier but upstream of the water return location that is accessible to migratory fish and resident brown trout.

2.2.5 Mechanical damage

Anadromous fishes, such as salmonids (*Salmo* and *Oncorhynchus* spp.) and shads (*Alosa* spp.), and catadromous fishes, such as eels (*Anguilla* spp.), must pass from fresh water to the sea as part of their life cycles, and may encounter hydroelectric dams and turbines (Cada *et al.,* 2006). Many non-diadromous species also make long-distance migrations through freshwater systems and may pass hydroelectric facilities. It should be noted that many of the issues raised are based on the available literature, mainly reporting on large turbines at dams, nevertheless the basic principles remain the same for any given turbine. A major issue for such fishes at hydroelectric facilities is injury to and mortality of eggs, larvae, juveniles and adult fish that pass through the turbines (Cada *et al.,* 2006). Fish passing through intakes that have no device for protection can be stressed, injured and sometimes killed (Hartvich *et al.,* 2008). Similarly observations of fish passage through turbines at 23 bypass- or mill stream-type small hydropower stations over a 50 day period (Kubecka *et al.,* 1997) found considerable numbers of brown trout, perch (*Perca fluviatilis* L.), rainbow trout (*Oncorhynchus mykiss* (*L.*)) *and* eels of a range of sizes were killed by turbines. Similarly the cumulative mortality rates of juvenile salmon passing 23 small-scale hydro-plants in France were as high as 64% (Larinier, 2008). Abernethy *et al.* (2002) reported the change of pressure experienced during passage through a Kaplan turbine damaged and in some cases killed certain fish species. Such mortality appears to be species specific. Moursund *et al.* (2003) concluded that Pacific lamprey *Entosphenus tridentatus* (L.)) transformers did not sustain any injuries or mortalities whilst passing through a Kaplan turbine under those same pressures or shear stresses that were found to injure or kill juvenile salmon. This is perhaps explained by their lack of certain body parts, such as swim bladders, that are more susceptible to damage and their incredible flexibility.

There is great potential, however, to preserve and/or restore fish stocks, whilst simultaneously maintaining a valued source of renewable electricity, if the survival of turbine-passed fish can be increased (Cada *et al.,* 2006). Injuries and mortalities commonly found amongst turbine-passed fish can result from several mechanisms (Figure 2.10), including extreme and rapid pressure fluctuations, turbulence, grinding, strike, cavitations and shear stress (Odeh, 1999a).

Figure 2.10 Schematic diagram showing locations within a turbine system where fish injury mechanisms are believed to occur (adapted from Odeh, 1999b). 1 Increasing pressure; 2 Rapidly decreasing pressure; 3 Cavitations; 4 Strike; 5 Grinding; 6 Shear; 7 Turbulence.

Water pressures within turbines can dramatically increase and suddenly drop in a matter of seconds. Under normal circumstances in an unregulated river, water pressure changes vary between one and two atmospheres, whilst pressure changes in turbines are of the order of four atmospheres (Becker *et al.,* 2003). Similarly dissolved gases are likely to exceed 130 percent of saturation below and above dams compared with levels below 100 percent saturation given no impoundment (Becker *et al.*, 2003). Hadderingh & Bakker (1998), Boubee & Williams (2006) and Larinier (2008) reported forces can be severe and brutal enough to cause tearing and/or bruising of tissues, de-scaling and decapitation (Normandeau *et al.,* 1996). Eels are at particular risk because of their body shape (Larinier & Travade, 2002), and hydropower stations have been cited as contributing to, if not being partly responsible for, declines of *Anguilla* species worldwide (Prosek, 2010). Richkus *et al*. (2003) reported mortality rates of between 15-30% for downstream-migrating eels, and considerable mortality has been reported in other schemes (Therrien & Bourgeois, 2000). Conversely, Kubecka *et al.* (1997) failed to capture any fish in drift nets installed in the outflows of turbines of small hydropower schemes, suggesting that fish can possibly more easily avoid intake zones at small schemes compared with larger hydroelectric plants. Filtering screens with apertures of approximately 10-12.5 mm are commonly used at most small hydropower schemes in Scotland to prevent entrainment of larger fish; though screens were designed for salmonids only and not eels.

Mortality amongst fish that pass through turbines is not always directly related to physical damage. Concerns have also been raised about indirect mortality. Fish may experience low levels of physical stress during turbine passage, but subsequently die in response to increased susceptibility to predation or disease (Cada, 2001). This is, however, less well documented and therefore it is unclear if this is a significant factor contributing to mortality associated with turbine passages.

There has been little research on downstream passage survival rates, with most work restricted to temperate rivers in North America, Russia and northern Europe (Table 2.1). Two measures are used to express survival rates: downstream passage success (DPS) and downstream passage mortality (DPM). While DPM provides a measure of the number of fish that are killed during passage through hydroelectric facilities, DPS measures the percentage fish that survive the passage. However, DPS does not account for fish harmed during passage and that later die from their injuries, or that are unable to complete their life-cycles. Therefore, survival rates in the medium to long-term may be compromised.

Measurements of DPM are only available for a few studies. These limited data show a great range in mortality rates of juvenile salmonids passing through Francis turbines (5-90%) and Kaplan turbines (5-20%), and over spillways (0-37%). DPS rates also vary greatly according to the location, design and operating conditions of particular dams and the fish species passing through the hydropower plant. Behrmann-Godel & Eckmann (2003) and Winter *et al*. (2006) reported increases in mortality, due to injuries acquired when passing through hydropower turbines, and alterations in eel behaviour (i.e. a delay in migration past the structure) associated with the downstream migration of silver eel at low-head hydropower schemes.

Table 2.1 Summary of reported downstream passage success in river basins. DPS – Downstream passage success; DPM – Downstream passage mortality. NA – Not available, # – small-scale hydro-electric plant.

Location	Species	Life stage	Pass type	DPS %	DPM %	Reference
NA#	Salmonids	Juvenile	Francis Turbine		5-90	Larinier (2001)
NA#	Salmonids	Juvenile	Kaplan Turbine		5-20	Larinier (2001)
Columbia River, USA	Salmonids	Juvenile	Spillway		0-37	Larinier (2001)
Connecticut River, USA	Salmonids	Smolts	Louver screen	97		Larinier (2001)
	Clupeids	Juveniles	Louver screen	86		
USA / France	NA	NA	Surface by-pass	60-85		Larinier (2001)
Ural River Russia	NA	Juvenile	Natural by-pass	81.5		Pavlov (1989)
Kuban River, Russia	NA	NA	Floating boom	67		Pavlov (1989)
Russia	NA	NA	Hydraulic screen	55-100		Pavlov (1989)
Ice Harbor Dam, Snake River, USA	NA	NA	Bypass	3-17		Goodwin *et al.* (2006)
	NA	NA	Spillway	78-89		
	NA	NA	Turbine	10-21		
Wanapum Dam, Columbia River, USA	NA	NA	Bypass	3-17		Goodwin *et al.* (2006)
	NA	NA	Spillway	0- 61		
	NA	NA	Turbines	32-91		
Lower Granite Dam, Snake River, USA	NA	NA	Bypass	0-78		Goodwin *et al.* (2006)
	NA	NA	Spillway	5-89		
	NA	NA	Turbines	8-96		
Gave de Pau, France#	Salmonids	NA	Bypass*	34-100		Larinier (2008)
River Mosel, Germany#	*Anguilla anguilla*	Adult	Turbine			Behrmann-Godel & Eckmann (2003)
River Meuse, Netherlands#	*Anguilla anguilla*	Adult	Turbine			Winter *et al*. (2006)

It is therefore difficult to draw any general conclusions about survival rates for particular methods of fish passage, but considerable losses can be incurred and the cumulative effects of passing many hydropower facilities are likely to be highly detrimental to the survival and sustainability of fish populations. This is the case for both small-scale, run-of-river schemes and large dam schemes.

The most common cause of fish mortality passing through turbines is injuries by blade strikes and also pressure stresses. There is a close correlation between the length of fish and the probability of mortality caused by a blade strike. Fish 50-cm long have a 40% chance of being killed by a blade strike (Halls & Kshatriya, 2009). This rises to nearly 100% for fish that are longer than 1 m (Figure 2.11). Therefore, older fish and larger species, especially eel, are more vulnerable than young fish or smaller species.

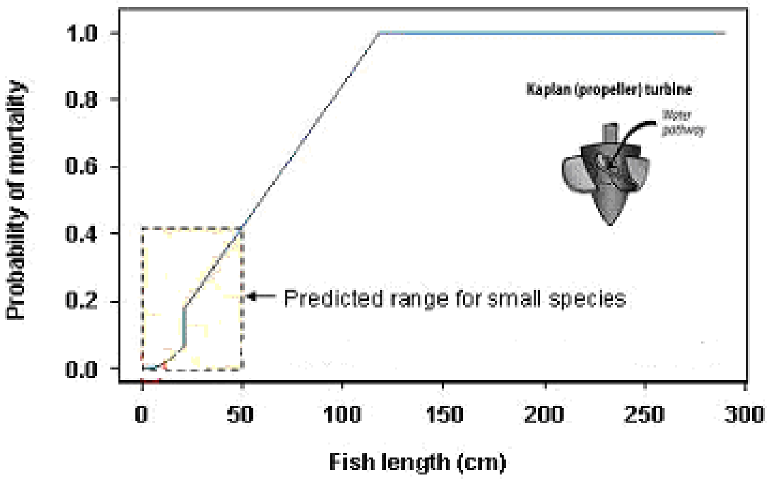

Figure 2.11 Relationship between fish length and the probability of mortality due to a blade strike. Source: Halls & Kshatriya (2009).

Bell & DeLacy (1972) further documented the effects of moving downstream through weir structures with impacts including abrasion, embolism, eye damage and death. Fish moving over spillways can be injured or killed if the design of the spillway or downstream bypass facility do not account for fish passage. Larinier (2001) reported that significant damage occurs to fish when the impact velocity of fish on the water surface below spillways exceeds 16 m s^{-1}. This velocity is reached by water falling from a height of 13 m.

Injury and mortality rates increase rapidly with greater fall heights and reach 100% for falls of 50-60 m, although it should be noted that such heights are not experienced in run-of-river hydropower schemes in the UK. If the flow is too strong fish may be unable to avoid collisions with energy-dissipating structures (such as concrete détentes) and flow deflectors, may suffer abrasion against spillway walls and floors if water is too shallow, and may suffer 'gas bubble disease' if plunge pools are too deep. SEPA's guidance states that the depth of a plunge pool must be at least 1/3 of the height of the vertical drop or 1 m, whichever is the smallest. Furthermore, turbulent flow in spillway basins can disorientate fish, slowing their downstream movement and exposing them to predation (Larinier, 2001).

Even if fish survive passage over spillways or through turbines, any injuries suffered during passage are likely to reduce their chances of survival. Also, when confronted with a structure such as a hydroelectric dam, downstream migrants spend time searching along the headrace, presumably for an unobstructed pathway downstream (Haro *et al.,* 2000). Eels that are unable to find a pathway tend to return upstream, often to where they were residing previously (Watene *et al.,* 2003), resulting in a net loss in the reproductive potential of the population.

2.2.6 Weir pools

One area that has received little attention with regards to the impact of hydropower development is the downstream weir pool. Weir pools can be important for spawning and development of several riverine fish species, such as barbel, dace, chub, bullhead, and stone loach, and as a habitat for macrophytes and invertebrates. The power of the water passing over the weir can maintain the weir morphology and clean gravels that are used by the aforementioned species for spawning. In some large slow flowing rivers, these gravels may be the only spawning habitat for several kilometres of river and thus make significant contributions to the fishery and wider ecology for a distance downstream. Any modification to the flow dynamics of the weir brought about by a hydropower scheme could affect achievement of good ecological potential (or status if not considered a heavily modified water body). The problem arises because a turbine situated on, or immediately adjacent to, a weir may discharge water into the weir pool, but the flow pattern and energy dissipation will have been changed. It is therefore recommended that the hydraulic conditions in the weir pool are maintained. In particular it is important the flood flows that create the appropriate weir pool morphology and the intermediate (Q_{s0} - Q_{50}) flows that maintain the gravels in suitable condition are protected.

2.2.7 Conservation species

Conservation species such as Atlantic salmon and eels are currently protected under European legislation and could potentially be impacted by the run-of-river schemes within this study. Indeed, while there are many other species of high conservation importance which may be impacted by schemes, such as the freshwater pearl mussel, Atlantic bryophytes and lampreys, they were not the study species within this investigation and therefore were not the focus within this review.

The Wildlife and Countryside Act, 1981, was until recently, the primary legislative tool for protecting UK heritage. This Act protects a number of named plants and animals (Table 2.2), as well as establishing Sites of Special Scientific Interest (SSSIs). SSSIs are considered the essential building blocks of Scotland’s protected areas for nature conservation; many are also designated as Natura sites as either Special Protection Areas (SPA’s) or Special Areas of Conservation (SAC’s). These are the areas of land and water that Scottish Natural Heritage (SNH) considers to best represent our natural heritage, its diversity of plants, animals and habitats, rocks and landforms. The Nature Conservation (Scotland) Act, 2004, however, has more recently strengthened these provisions and placed a statutory duty on all public bodies to further the conservation of biodiversity. In Scotland, a total of 441 running and standing water bodies are included within the SSSI series (Bean & Thin, 2008). Although there are a number of international nature conservation and water resource management agreements and directives, European directives (Habitats and Water Framework Directives) are now considered to be the primary driver for improved conservation measures within Scotland and the UK, relevant to the production of hydropower schemes. The Habitats Directive, together with the 1979 Birds Directive, provides a framework of sites, collectively known as “Natura 2000”, to protect the most seriously threatened species and habitats. These include Atlantic salmon, brook, sea and river lamprey, and the freshwater pearl mussel.

Table 2.2 Features with freshwater affinities within the Scottish SSSI series. (*) denotes interest features for which it is difficult to identify those species that use only freshwater habitats. (Bean & Thin, 2008).

SSSI interest feature	Number of SSSIs containing this interest feature
Non-vascular plants (bryophytes and lichens)	16
Vascular plants (macrophytes)	26
Freshwater molluscs (inc.f/w pearl mussel)	6
Freshwater invertebrates (species and assemblages)	12
Dragonflies	27
Amphibians	8
Fish (Atlantic salmon, brook, river and sea lamprey, Arctic charr, powan, sparling and important fish assemblages)	18
Aggregations of breeding birds*	424
Aggregations of non-breeding birds*	404
Bird assemblages*	160
Otters	8
Rivers and streams	10
Standing open waters	146

Whilst there are negative impacts, the linkage between hydropower and fisheries in some instances has provided some arguably natural heritage gains. In addition to providing the basic infrastructure, a high proportion of fish counters currently operational within Scotland are associated with (large-scale) hydropower schemes. This is not a mandatory requirement for hydropower operators, yet this has provided valuable, fisheries-independent information regarding the status of migratory salmonids.

2.2.8 Impacts during construction phase

The initial construction of hydropower schemes may cause an array of environmental impacts such as increased sedimentation, physical damage to the landscape and noise pollution. Physical damage to the environment may include ground clearance leading to the removal of vegetation cover and/or trees if located in forested areas, trenching to bury pipelines, blasting and grading. Pedestrian traffic, noise and visual pollution may not be a cause for concern regarding high-head schemes due to their location but could be problematic for humans for low-head schemes. During the construction of new access roads and pipelines, top soil/vegetation is removed, which has potential ecological consequences including reduction of plant diversity and wildlife habitat. Sediment disturbance and run off can ultimately lead to weathering of newly exposed soils that could potentially cause leaching and oxidation; the release of new chemicals into the rivers, which can harm fish populations. Amongst the impacts of silt pollution are reduced water quality, increased risk of flooding due to the blocking of culverts and channels and as mentioned above damage and cause mortality of aquatic species due to smothering and suffocation. As a consequence, precautions should be taken to mitigate potential impacts (see later section) such as the use of silt traps.

2.2.8.1 Intakes and outfalls

The following section is an overview of the Intakes and Outfalls Guide (SEPA, 2008).

The construction of intakes and outfalls is not without risks. There are many potential impacts as a consequence of the changes they have upon sediment transport, river flow and substrate composition. The first concern relates to their construction, causing direct loss of bank side habitat. Overhanging trees and shrubs can be lost, which provide cover and shelter for many juvenile fish and riparian habitat is also a prime source of food for an array of aquatic species. Increased sedimentation is also a risk as mentioned previously; increased sedimentation can result from mechanical and engineering work, but additionally a build up of sediment in front of the intake is a potential impact. In addition reduced flow will hinder sediment transport to downstream reaches and potentially affect the deposition of sediment in upstream reaches. Entrapment of fish is also a possibility if appropriate screens are not in place.

2.2.9 Cumulative impacts

Currently, legislation in England and Wales (Schedule 2 of the Town and Country Planning Regulations 1990) only requires schemes of >500 kW installed capacity to provide an Environmental Impact Assessment (EIA). Similarly, in Scotland and Northern Ireland, schemes <100 kW do not require an EIA, unless they are located in conservation or heritage sensitive areas (e.g. (SSSI), (SAC) or (SPA)). Consequently, because of their marginal economic profile, many small schemes are submitted with no, or a rudimentary, environmental statement, and little consideration for mitigation of any likely impacts. This said, it should be taken into account that Scotland provides guidance on mitigation that applies to all schemes and as such screens are part of the criteria for almost all new schemes. Whilst single hydropower schemes are considered less likely to have catastrophic impacts on fish community structure and dynamics, or compromise the ecological status of rivers as a whole, the cumulative effects of more than one scheme in a river or catchment can potentially be more ecologically damaging.

Paquet & Witmer (1985) summarised the occurrence and variety of cumulative impacts, which include delays in fish migration, fish mortality at impoundments, losses of fish spawning and rearing habitats, and losses of adult fish due to blocked migration, as well as issues with water quality, invertebrates, aquatic plants and predators on fish. They defined cumulative impact as “the total iterative impacts over time, i.e. the sum incremental, synergistic effects on fish and wildlife populations and habitats caused by all current and future actions over time and space.” Thus, if impacts are cumulative at a series of schemes, fish needing to traverse a catchment to spawn or migrate downstream could experience considerable losses or impairment, leading to deterioration of fisheries. For example, if a single proposal causes 10% mortality of salmon smolts through a turbine, over 40% of the downstream migrating stock could be lost over a series of five such schemes. Couple this with a 95% upstream migration passage rate at each scheme and the total net loss to the fishery could be in excess of 60%. To support this simple analysis, Gowans *et al.* (2003) tracked 54 adult salmon through four fish passes and an impoundment on the River Conon, as an example of upstream passage. Percentages of fish passing individual obstructions ranged from 63 to 100%. Individuals were delayed by up to 52 days at one pass, and only four salmon actually reached the spawning areas, reflecting the cumulative effect of multiple barriers. Although this case was based on larger hydropower impoundments, it illustrates the effects barriers can have.

Run-of-river barriers should not present such severe impacts, but if flows are reduced significantly in long river reaches as a result of flow diversion through the turbines, there is the potential for fish not being able to negotiate long lengths of river or bypassing even small barriers such as weirs. In this context, it is noted that the presence of existing weirs may already impact in fish migration and it is the reduced ability of fish to bypass depleted reaches and barriers as a result of flow diversion that is the issue. It also highlights the uncertainty surrounding fish passes and that fish may encounter potential problems locating a fish pass before successfully moving upstream. For example, Lucas *et al.* (2009) investigating the distribution of, and access to, fragmented spawning habitat for river lamprey in relation to the presence of potential low-head barriers concluded that river lamprey were restricted in their ability to use areas of river upstream, although it is possible that the lamprey deliberately halted their migration upstream as spawning areas were present below the barriers. Nevertheless, there was a reduced number of river lamprey at successive spawning sites upstream of consecutive barriers.

The current proliferation of schemes in the UK can be compared with the scenario enacted in the United States from 1970, following changes in legislation. A conclusion from a symposium on small hydropower and fisheries in 1985 found that “in its separate pieces small hydro does not have adverse effects on the environment. But taken as a whole, thousands of small hydro plants can go far towards depriving this country of much of the ecological, cultural, and aesthetic properties of our streams and rivers” (Campbell, 1985). Larinier (2008) also concluded from his experience in France that it is poor management practice to plan the construction of more than a very limited number of small-scale hydropower stations on a river, and Rizzo (1985) concluded that “we must get away from the present method of evaluating multiple hydropower projects on a project by project basis”.

2.3 Responses to issues – mitigation measures

2.3.1 Introduction

The previous section has highlighted a number of potential impacts of run-of-river hydropower schemes on fisheries in rivers. The main impacts are associated with the impoundment structures impeding migration of fish, loss of ecological integrity in depleted flow reaches, mechanical damage to fish passing thorough turbines, and to a lesser extent modification of the downstream weir pool. The scale and intensity of these impacts depend very much on the position of the schemes in the river, e.g. high versus low-head schemes in upper and lower reaches of rivers, the species and size of fish associated with the section of river impacted and modifications to the hydrology of the river. It should also be recognised that whilst these impacts vary for individual schemes, they are not exclusive to the scheme, and cumulative impacts may arise from multiple schemes in the same catchment. It should also be noted that many low-head schemes are developed on existing weirs that may already impose impacts on fish populations.

In response to concerns over the impacts of run-of-river hydropower schemes, a suite of measures has been developed by the hydropower industry practitioners, fisheries managers and other stakeholders to mitigate or ameliorate the impacts, but their effectiveness is relatively unknown as the few studies that have been published have proven to be inconclusive. There are considered to be four categories of environmental and social mitigation measures (modified from Trussart *et al.,* 2002).

1. Impact avoidance measures – implemented at the initial planning and design stage of the project to ensure no impact is likely to occur;
2. Mitigations measures – used to reduce or eliminate a source of impact completely;
3. Compensation measures – compensate for those impacts that cannot be mitigated; and
4. Enhancement measures – increase environmental or social benefits beyond those affected directly by the scheme.

Amongst the mitigation options are: 1) construction of a fish-way at the turbine outlet, so fish can find an upstream route; 2) regulation of spill flows to secure successful attraction and passage efficiency of the bypass; 3) establishing environment flow standards; 4) downstream bypass channels; 5) screening; and 6) improved design of the turbines. These are discussed in the Sections 2.3.2-2.3.7.

2.3.2 Fish passes/ fish ways

Irrespective of the design of hydropower schemes, they tend to be associated with some sort of impounding structure. In high-head schemes a small impounding weir is usually constructed to hold back sufficient depth of water to flood the off-take structure (see Figure 2.3), whilst low-head schemes are usually associated with an impounding weir to provide the volume of water required to drive the turbine (see Figure 2.5). In many cases impounding weirs on high-head schemes do not create a barrier as they tend to be immediately upstream of a steep gradient in the river profile such as a waterfall, which acts as a natural barrier to migration. However, in some cases this can be upstream of a river section with a steep gradient that is passable by fish and may thus impose a threat to free movement. Indeed, there are also weirs, which although small, can act as a barrier because the compensation flow over them is small; under these circumstances fish passes are necessary. At low-head schemes, the barrier may be an obstruction and thwart or delay fish passage unless a specific bypass facility is integral within the barrier. The ability to negotiate the structure will depend not only on the topography of the barrier but also the flow regime and how it has been modified by the hydropower (or other flow regulation) development. Some obstructions may only be passable during periods of high-flow or at a particular range of flows. O'Connor *et al.* (2005, 2006) found increased numbers of fish moving upstream over weirs as flow increased. This is perhaps unsurprising as sufficient water depth associated with an increased flow is needed for fish to move upstream and specifically to leap over barriers such as weirs. Low water depth means fish are not fully submerged and thus may not be able to swim efficiently or pass obstructions such as weirs. If the flow over the obstruction is reduced either by diverting water through a different channel or through a turbine on the impoundment, fish passage may be delayed or even prevented. Without mitigation, this could potentially threaten the long-term survival of natural salmonid and other migratory fish populations (Lundqvist *et al.,* 2008), leading to failure to achieve the WFD objectives. There is therefore a clear need to enhance fish passage where migration pathways are impaired. This is accommodated under the Salmon and Freshwater Fisheries Act 1975 and the Water Environment (Controlled Activities)(Scotland) Regulations 2011. It should be noted that as part of the *Green Hydro* certification, the provision of fish passage facilities at new, or re-licensed, hydropower schemes is insisted in many countries, including France and Germany. See Section 2.4 for UK fish passage requirements.

The best, when circumstances allow, and increasingly popular way to re-establish longitudinal connectivity, is to decommission (i.e. remove) dams, weirs or other obstacles; when run-of-river schemes are not permitted on the existing feature. This option is gaining increasing prevalence in North America and Europe when obstacles have ceased to serve their original purpose, when licences have expired, when repairing and retrofitting with fish passage facilities is no longer economically viable, or where ecological aspects are put above economic considerations. This option is not appropriate for many barriers to migration, especially at low-head schemes where the impounding structures have heritage value and may restrict the mobilisation of contaminated sediments, but could also be at odds with commitments towards renewable energy. Consequently, fish passes are suggested to be amongst the best methods to mitigate the effects of river obstructions associated with run-of-river hydropower schemes (Lucas & Frear, 1997; Lindmark & Gustavsson, 2008; Godinho & Kynard, 2009), thus allowing migratory fish to reach their required spawning grounds (Jungwirth, 1996; Knaepkens *et al.,* 2005).

According to the Environment Agency’s Good Practice Guidelines (EA, 2009a), a fish pass is required under the Salmon and Freshwater Fisheries Act, in waters frequented by salmon and sea trout if:

- a new impoundment is constructed, or
- if an impoundment is rebuilt or reinstated over half its length, or
- if an existing impoundment is altered physically, or
- as a result of flow reduction so as to create an increased obstruction.

Where an existing impounding structure is partially passable, removing flow from it to a hydropower scheme will in most circumstances reduce passage for fish. It may prevent passage altogether, or more likely reduce the window of opportunity for fish to pass. Thus, as a condition of the abstraction licence, impoundment licence or Flood Defence/ Land Drainage consent, a fish pass is required if the species of fish present will experience increased difficulty completing their life cycles as a result of the hydropower installation, especially if this will lead to a deterioration in ecological status.

Other legal obligations may be applied where sites or species affected have nature conservation designations, e.g. under Habitats Directive, SSSI or are the subject of European conservation plans e.g. eel as regulated under the EU Eel Directive. The Water Environment (Controlled Activities)(Scotland) Regulations 2011 also allows SEPA to require a fish pass to be installed at any location not just where migratory fish are present.

If a fish pass is required in England and Wales, the design must be approved, e.g. by the Environment Agency (Armstrong *et al*., 2010). Importantly, where a fish pass is already present, or where a fish pass is provided by the scheme, the EA (2009a) recommend the downstream fish pass entrance and the discharge from the turbine(s) are co-located to enhance attraction to the vicinity of the pass, helping alleviate one of the major problems with fish pass functioning (see Section 2.3.2.2). Under these circumstances, an attraction flow through the fish pass of at least 10% of the turbine flow is recommended, but this may need to be considerably higher to support effective functioning of the fish pass.

2.3.2.1 Fish pass structures

Fish-ways are passages designed to dissipate the energy in the water to aid fish ascent at migratory obstructions (Clay, 1995). There are four general requirements for fish passes (Cowx & Welcomme, 1998) (see below), but their effectiveness can vary depending on certain factors such as the fish species in question, water head, design and size of river. These are:

- Sufficient capacity to allow large numbers of fish to pass in a limited timeframe.
- Adapted to the swimming capacities of the fish, noting different fish species swim at different speeds and the pass should be designed to permit passage of slower swimming species and all life stages. The velocities and turbulence conditions must be adapted to the capacities of the target species. The dimensions of a pass must be chosen in relation to the body size of the biggest individuals that are expected to occur. Each fish pass must be designed in such

a way as to function satisfactorily under varying flow conditions, within reasonable limits. These flow conditions must be considered in assessments of environmental flows.

- Permanently functional – the pass should be permanently operational and should be able to function over the different flow regimes of the river. All fish passes need regular maintenance to clear accumulated trash and ensure that flow is not obstructed. Often guidance suggests that a fish pass only needs to be operational when fish would be expected to use it, but this does not acknowledge that most fish species potentially migrate throughout the year and different species may have peak activities at different times of the year.
- Positioning of the entrance – the entrance should be readily accessible to migrating fish and thus should be positioned in the main stream of the river. To improve the likelihood of fish finding the entrance, it should where possible be co-located with hydropower turbine outfall to provide attraction flows.

The best options for mitigation are those passes that simulate as closely as possible the features of the natural water course. Fish passes that are most effective provide sufficient attraction flows (Trussart *et al.,* 2002) and flows through the pass at critical times of migration. Consequently, flow is crucial to the efficiency of a fish pass, and a hands-off flow should be set at a level that allows fish to ascend the fish pass at all times. The following sections describe several options that are available. It should be noted these options are really only available at barriers associated with low-head schemes and bypassing the small off-take impounding weirs in high-head schemes. It is not within the scope of this research to review fish passage design; however an overview is provided in the following sections while detailed information on the design and selection of fish passes are provided in the Environment Agency's Fish Passage Manual (Armstrong *et al.*, 2010) and SEPA Guidance on Passage Selection (SEPA, 2010a).

The following sections provide an overview from the two fish passage manuals named above.

Nature-like passes

Fish ramps - Fish ramps are hard engineered structures where natural materials such as boulders are used to create slopes that allow a great variety of fish and invertebrates to ascend or descend the river. The use of such ramps is limited to low obstacles and are ideal for the impounding weirs on high-head schemes. Strategic placement of boulders can direct currents and provide shelter for fish as they ascend the structure. As in other fish passes, the velocities and turbulence conditions on the ramp must be adapted to target species. The width of the structure and its generally low slope means that also downstream migrants have a good chance of survival. Ramps have numerous advantages in that they are more aesthetically appealing, they provide a varied series of habitats and can be placed into the main channel of the river with no or little requirement for extra land. Being constructed in the main channel, fish have little difficulty in finding the entrance, but they tend to have high maintenance requirements as they are susceptible to clogging.

Bypass channels - Bypass channels are long channels around obstacles that begin downstream of an obstruction and end upstream of the turbine intake. They have the advantages of blending well into the landscape (Jungwirth, 1996) and providing additional habitat for spawning and juveniles of a wide range of species (A. Butterworth, pers. comm.). They can be passed upstream and downstream by a large variety of fish (Eberstaller *et al.,* 1998; Calles & Greenberg, 2007). Unfortunately, bypass channels usually require a large amount of land, because their low gradient from less than 2% to a maximum of 5% necessitates they are often very long to overcome a given weir height (Larinier, 1998) Furthermore, they are often expensive to install. They also require a continuous flow to be diverted from the river upstream to function (Jungwirth, 1996), and this can lower the water available for hydropower generation, which may be a problem if economic returns of the hydropower scheme are marginal. They are also very sensitive to fluctuations in water levels (headwater and downstream) and sometimes have to be connected to the river by technical sections at the upper and lower ends. As for all passes, the entrance has to be in an optimal position for fish to find it without problem. Godinho & Kynard (2009) reported perhaps a downfall of nature-like fish-ways was the entrance must be located several kilometres downstream of the dam, which would likely decrease the chances of fish finding it; on the other hand, the entrance to a technical fish-way can be positioned in the dam's tailrace. They have a reduced tendency to clogging thereby lowering the cost of maintenance.

In reality, bypass channels are unlikely to be used on high-head schemes due to the small weir, but may be important for low-head schemes.

Technical fish passes

Pool-type pases and baffle passes - Technical fish passes are hard engineered structures usually of the pool-and-weir, pool, or baffle type, intended primarily to assist fish migrating upstream. Many different designs are available with different types of cross walls and pool dimensions. The design (i.e. pool size and drop between pools) must be adapted to the swimming capacities of the target species. Pool-and-weir passes are one of the oldest designs and have proved their value for strong swimming species as well as some bottom migrating species (if there are bottom orifices or vertical slots and if good maintenance is done) (Armstrong *et al.,* 2010). Depending on the design (especially the size of the openings), their flow requirements can be relatively low but they require high maintenance as the orifices and slots in the cross walls are prone to clogging. A special type of pool-and-weir pass is the vertical slot pass, which proved efficient for many different species (Armstrong *et al.,* 2010). Vertical slot passes may have higher flow requirements as a function of the slot size. Baffle-type passes, e.g. the Denil pass, have the advantage that they can be used on relatively steep slopes and thus require limited space. They are more easily retrofitted to dams lacking fish passes than the pool-type passes, but they can be used by a smaller range of species because they are more selective as a result of the particular flow conditions (Armstrong *et al.,* 2010). This has led to their efficiency being questioned in relation to the behaviour and swimming abilities of migratory species (Larinier, 1998). Nevertheless, Denil fish passes can be negotiated by species such as chub, bream (*Abramis brama* (L.)), bleak (*Alburnus alburnus* (L.)) and European eel if designed correctly (Baras *et al.,* 1994). Larinier (1998) reported visual references were of crucial importance; such that lower velocities should occur along a smooth wall (floor or sides), to guide the fish. An advantage is that they do not need high discharges. They have disadvantages in that they are intolerant of variations in headwater level and they are easily disturbed by clogging with debris requiring high maintenance.

2.3.2.2 Problems with fish passes

Damming and regulation of rivers have undoubtedly decreased the abundance of migratory fish (Ugedal *et al*., 2008). Although fish passes are seen as a potential mitigation measure to address the problems arising from the impoundment barrier constructed as part the hydropower scheme infrastructure, several problems have consistently been found that are typical of fish passage operational success. These include appropriate fish pass design to enable all fish species to utilise the facility successfully, attraction to the entrance, appropriate flows through the fish passes and provision for downstream migration (Cowx, 1998; Armstrong *et al.*, 2010).

Calles & Greenberg (2007) and Kemp *et al.* (2008) voiced concern that fish-way design is historically biased. In the early days, many fish passes were intended only to facilitate the upstream migration of economically and commercially important species, primarily adult salmonids such as salmon and trout, whilst smaller fish and species of lower commercial value were generally ignored. Non-salmonids, however, differ greatly in their abilities to negotiate physical and hydraulic barriers, consequently they have limited capacity to utilise passes designed for salmonids (Knaepkens *et al.,* 2006). Stuart & Berghuis (2002) suggested that many older pool-type fish-ways did not provide effective conditions for the upstream migration of juvenile catadromous fish (e.g. eel) and Knaepkens *et al.* (2006) found that a pool-and-weir fish pass in the regulated River Laarse Beek (Belgium) was totally unsuitable for upstream migration of bullhead and only enabled 8 and 29% of tagged perch and roach (*Rutilus rutilus* L.), respectively, to bypass the barrier. Excessive water velocity in the fish pass was suggested to be responsible for the failure of bullhead to use the pass. The leaping and swimming abilities of many cyprinid species, such as barbel and bream are also, for the most part, comparatively poor. These differences highlight the importance of fish pass design to suit an array of species and furthermore the need to understand all fish species' flow requirements. In most cases, an optimal solution to mitigate migration of fish species hindered by artificial obstacles, such as dams and weirs resulting from hydroelectric development, would be bypass channels that resemble natural lowland rivers with a comparatively flat gradient and a high morphological, current and substrate diversity are a viable alternative (Jungwirth, 1996).

Although there will always remain some uncertainty regarding the success of bypasses, the correct design and layout can improve the chance of success, but this will alter depending on the fish species of concern. Bypasses for salmonids are generally located nearer the water surface as they tend to swim in the surface waters during their migration (Gosset *et al.,* 2005). Conversely, it has been suggested that bottom bypasses are more appropriate for eels, given their benthic behaviour (Gosset *et al.,* 2005), although Watene *et al.* (2003) suggested the entrance to bypasses for eels should be positioned approximately 1 m below the water surface, justified by the knowledge that migrating eels travel close to the water surface. These conflicting results perhaps suggest that advantages may be gained by better located bypasses of larger diameters and multi-bypass systems (Boubee & Williams, 2006). Overall, however, bypass efficiency has been suggested to depend upon shape, location in relation to trash-racks and hydrological conditions near the entrance (Larinier & Travade, 1999; Guiny *et al.,* 2003). Furthermore, the bypass entrance should ideally be at least 30-60 cm wide and preferably extend to the full channel depth (Turnpenny *et al.,* 1998). The design and location of the fish-way entrance is, therefore, fundamental to its success.

Problems can also arise in the vicinity of fish pass entrances because flows through the pass are ineffective at attracting and ensuring successful upstream migration (Bjornn & Peery, 1992). Banks (1969) reported that when approaching a channel divergence that has no other directional clues other than flow, fish (specifically salmon) will inevitably be influenced by the greater volume and higher velocity of water when travelling upstream. This was supported by Weaver (1963) who found steelhead (anadromous rainbow trout), chinook and silver salmon choose the higher velocity when presented with two alternatives. Consequently, this behavioural characteristic should be used to attract fish to the entrance of the fish pass (Katopodis, 1990). This is particularly important when associated with a hydropower plant because the discharge from a fish pass is usually considerably smaller than that through the turbines (Laine *et al.,* 2002). As a consequence, the discharge from the turbine tailrace can dominate and attract the fish away from the bypass route, creating problems for the fish to even find the entrance to the fish pass (Arnekleiv & Kraabøl, 1996; Lundqvist *et al.,* 2008). Linlokken (1993) noted similar behaviour in grayling and brown trout, and Thorstad *et al.* (2003), Laine *et al.* (2002) and Rivinoja *et al.* (2001) found salmon were reluctant to enter an unobstructed bypass channels as they were attracted to the higher water discharge from the turbine channel. This is further supported by the work of Karppinen *et al.* (2002) who reported that upstream migrating salmon in the regulated River Tuloma, Kola Peninsula, were reluctant to enter the fish passes and occasionally even backed out having entered the fish pass as they preferred to seek their way in the stronger current, i.e. the tailrace and spillway discharges. It is, therefore, essential that there is a strong flow attracting fish to the fish pass entrance (Cowx & Welcomme, 1998) preferably with the entrance velocity equal to or greater than the main channel velocity (Turnpenny *et al.,* 1998). To support this argument, Aarestrup & Koed (2003) noted that as flow through a bypass channel increased the passage of salmon increased, but if the flow was too low the salmon entered the turbine outlet. Jungwirth (1996) hypothesised that if discharge through turbines was higher than the bypass discharge then more fish would be attracted to the turbine entrance, but the effectiveness of a bypass could be improved simply by releasing all of the remaining water through the bypass channel. This option may not always be acceptable and Turnpenny *et al.* (1998) suggested that at least 2% of the rated turbine flow should be used in the bypass channel; irrespective it is critical to create conditions that the fish do not avoid (Kynard, 1993).

Flow from the turbine tailrace can dominate the main river flow; therefore Clay (1995) highlighted the importance of the entrance to fish ways being positioned as close to the obstruction as possible, stimulating and enabling fish ascent from the tailrace. Positioning the entrance near the outlet of turbines ensures fish will be attracted to the main current channel increasing the chance of fish finding the bypass; where a turbine is located on a weir, the turbine outflow should be adjacent to the fish pass, although it should be recognised the success of a fish pass will depend on the hydraulics around the fish pass entrance and the proportion of flow through the pass, and these must be examined on a site by site basis.

An optimum design and location is not only essential to allow the fish to find the fish pass easily but also to prevent any increased and unnecessary stress (Clay, 1995). Mathers *et al.* (2002) reported that significant extra stresses, such as finding and negotiating a fish pass, multiple attempts to leap impoundments, and diversion into routes with no fish passage could potentially leave fish more susceptible to predation, poaching or stress induced diseases; fish may even completely fail to reach their spawning grounds. It has also been noted that the colour of the channel could be important to attract fish. Lindmark & Gustavsson (2008) examined fish passage of salmon and brown trout over a 3-week period in two consecutive years using a flow device that accelerated turbine tail water. They found considerably more fish (471 versus 57) passed through the channel in the second year when the channel was painted black, irrespective of the acceleration of the water velocity. Conversely, Turnpenny *et al.* (1998) reported that fish tend to avoid or resist entry into any form of bypass that does not admit light, suggesting open-topped bypass channels to be the favourable option.

Finally, when confronted with man-made obstructions, fish can alter their behaviour potentially leading to an increase in energy expenditure. Salmon smolts repeatedly move back and forth as they approach barriers, appearing to search for a surface outlet (Giorgi *et al.,* 1988), and when close to hydropower stations eels exhibit circling behaviour (Behrmann-Godel & Eckmann, 2003; Jansen *et al.,* 2007). Gosset *et al.* (2005) noted similar foraging behaviour with eels in the bay, ranging from 30 s to 14.25 days, either searching for a bypass or perhaps resting when hydrological conditions were unfavourable (low and/or no turbine discharge). This was also reported by Arnekleiv & Kraabøl (1996) during radio tracking studies of brown trout. Fish had to pass through a tunnel from an unregulated river to a regulated river that had significantly less water flow; a condition experienced in run-of-river schemes. The confluence between the regulated river and the outlet tunnel from the power station appeared to cause migration hindrance. Several fish wandered back and forth over a few hundred metres displaying what was described as “restless” behaviour. It was noted that when discharge along the regulated stretch was reduced to almost 20 m^3/s, this behaviour was heightened with some fish remaining in the area for a few days before completing their passage; 33 of 57 fish continued the migration along the regulated stretch. The remaining 24 either wintered in the area, continued further upstream when the autumn flood brought an increase in flow and others returned downstream after waiting at the tunnel mouth for more than 20 days (Arnekleiv &

Kraabøl, 1996). These “yo-yo” migrations, which could occur during times when fish are trying to negotiate a weir, such as those on run-of-river schemes, cause increased swimming, increasing energetic costs for the fish, which cannot be recovered; maturing anadromous salmon do not feed during their migration journey in fresh water. This leads to lower fat reserves, which in turn causes lower fitness of individuals during competition for mates and could potentially cause lower overwinter survival, resulting in negative effects on the populations (Lundqvist *et al.,* 2008).

There are also several misconceptions about the requirements for fish passes, due to poor understanding of migratory movements (Godinho & Kynard, 2009). Amongst the most important misconceptions is the belief that fish passes are only needed to aid upstream spawning migrations, but upstream migration can be used for other purposes such as foraging during the non-spawning season (Lucas & Frear, 1997; Godinho & Kynard, 2007; Godinho *et al.,* 2007). Similarly, there is equally the need for bypass facilities to support downstream migration of fish, especially to avoid damage and mortality associated with passing through turbines (see Section 2.2.5) or passing down spillways or weir faces. Minimising the fall of spillways is one means to reduce mortality; ogive or ‘ski-jump’ shaped spillways are preferable since this shape tends to minimise abrasion to fish. Ensuring a sufficient depth of water at the base of the dam barrier with no submerged baffles or rip-rap will also help to reduce damage and mortality rates. Provision of downstream bypass channels may also reduce mortality, provided entrainment and impingement can be prevented at the intake screen and the channel entrance is positioned so the fish can find the entrance.

A further issue that arises with respect to provision of fish passage facilities at hydropower schemes is that current fisheries legislation in England and Wales stipulates a fish pass only has to be constructed if salmon, sea trout and/or eel are compromised (A. Butterworth, pers. comm.), and similarly in Scotland, fish passes are designed almost exclusively for migratory salmonids (Bean & Thin, 2008). The Water Resources Act 1991 allows a fish pass to be made a condition on the grant of an abstraction licence should the Environment Agency consider it to be necessary. In January 2009, the Government published a consultation document with several proposals to existing legislation; these were proposals to enable the Environment Agency to require the installation of a fish pass or the placement of screens to facilitate the passage of all migratory and freshwater species. The Government confirmed it was “essential that migratory and freshwater fish have access to the full length of the water course so that they can complete their life cycle”; however, the Department for Business, Enterprise and Regulatory Reform (BERR’s) Better Regulation Executives identified that the fish passage measures were ones that could have significant impacts on businesses and that given the current economic and financial climate, it was decided to not proceed with the new proposals until at least May 2011. Furthermore, it has been suggested that if there is no spawning or nursery habitat upstream of a barrier, upstream passage of migrant fishes is not useful and an upstream fish-way is not needed. This is a very polarised and historical perspective of the importance of fish migration to meet obligations under the WFD. While there is currently no primary legislative power to make passes for non-salmonid fish species it is being considered in the review of the present legislation.

2.3.3 Flow management

Flow is a supporting element in WFD, except for high status. To assess the impacts resulting from the construction and operation of water abstraction and impoundments, environmental flows have been used based on a percentage deviation from a natural flow (Acreman *et al.,* 2009). Many methods have been developed for defining environmental flows, such as hydraulic habitat analysis, look up tables, desktop analysis and functional analysis (Acreman & Dunbar, 2004). Each method differs in approach; for example hydraulic habitat modelling is expensive to apply, but is more suitable for impact assessment at specific sites. Currently in the UK, there are two projects to set environmental standards for water resources; the first defines water abstraction limits (SNIFFER WFD 48; Acreman *et al.*, 2006, 2008) – this is restrictive and controls abstraction from a largely natural river flow and the second defines ecologically appropriate flow releases (SNIFFER WFD 82; Acreman *et al.*, 2009) – a controlled management tool for man-made flows from reservoir releases; both help maintain a healthy river ecosystem (Acreman & Ferguson, 2010). The outputs of these projects have been used to develop the UKTAG restrictive flow standards currently used as the basis for regulation (EA, 2009a; SEPA, 2010a). These environmental standards and conditions are fundamental to UK agencies that use this information to support their decisions on how much water can be abstracted.

UKTAG is the UK Technical Advisory Group on the Water Framework Directive. A group of experts from both conservation and environment agencies developed environmental standards and conditions to underpin the implementation of the directive. In April 2008, a report was published defining specific standards for an array of environmental conditions such as water flow, water quality and water levels, which were suggested by UKTAG, to best support healthy communities of aquatic plants and animals.

The setting of environmentally acceptable flows is still regarded as an evolving science, and new methods are still surfacing (Acreman & Ferguson, 2010). Under the Water

Resources Act 1991, the Environment Agency requires any abstractor removing 20 m^3/d from a river or stream to obtain an abstraction licence. SEPA operates a four-tier system under the Controlled Activities Regulations (CAR) based on the volume of water to be abstracted. Abstractions of <10 m^3/d do not require authorisation from SEPA as they are covered by General Binding Rules (GBR) and are considered low risk activities (SEPA, 2008). Abstractions ≥10 and ≤50 m^3/d require registration with SEPA, abstractions ≥50 and ≤2000 m^3/d require a simple license and abstractions ≥2000 m^3/d require a complex license (SEPA, 2008). Therefore high-head and low-head hydropower schemes often require licensing by the competent authorities; indeed even small-scale systems that use a water wheel to divert free-flowing water usually require an abstraction licence due to the volumes of water diverted. The Central and Regional Fisheries Boards of Ireland guidelines for hydropower development (Irish Fisheries Board, 2007) suggest for rivers within which fish migration takes place, flows in the bypassed channel shall be 50% of the upstream flow or 12.5% of the mean flow, whichever is greater. Furthermore, it is recommended that where a scheme is proposed in an area that contains important spawning or nursery areas for salmonids, non-salmonid fish or lampreys in the context of the catchment, no scheme should proceed because of the sensitivity of such areas to alteration of flows.

The hydrological regime of regulated rivers can be significantly altered resulting in increased occurrence of low flows, reduction in frequency and amplitude of high flows and/or occurrence of flows at unnatural times (Old & Acreman, 2006). The EU WFD requires all water bodies to reach good ecological status by 2015 and achievement of this target is greatly influenced by river regulation. A key approach to integrate freshwater management with ecological sustainability is the provision of 'environmental flows' (Arthington *et al.,* 2006). Setting of environmental flows is not a new approach in protection of the aquatic environment. The traditional method of setting compensation flow was criticised by Baxter (1961) as it was considered unrelated to biological need; they were based entirely on catchment water yield and at fixed rates. Baxter (1961) identified that there should be a variable compensation flow regime based on the entire range of seasonal needs of fish and the river. A residual flow recommendation based on maintenance of low flow conditions was also questioned by Tennant (1976) comparing it to "prescribing a person's all-time worst health condition as a recommended level for a portion of his future well being". Tennant (1976) discussed that although 10% of the average flow would provide minimum protection, 30% of the average flow was at a satisfactory level and 60% would provide excellent habitat. Others (Petts *et al.,* 1996, Poff *et al.,* 1997; Acreman & Dunbar, 2004) voiced similar concerns arguing that the concept of maintaining minimum "critical" flow is adequate for aquatic biota is both scientifically and environmentally flawed. They stated that a single minimum flow can create negative consequences for ecosystems as it does not represent natural flow fluctuations; all elements of a flow regime are important, including floods, medium and low flows. Although the basis of these comments relates to controlled flows from reservoirs it can be used as a learning tool about the impacts of such flow regimes. Kubecka *et al.* (1997) and Jowett (1997) concluded that schemes constructed on small rivers are more at risk from low flows than large and therefore require a higher proportion of average flow to sustain environmental protection. Thus setting of environmental flows should relate to the quantity, timing, duration, frequency and quality of water flows required to sustain freshwater and estuarine ecosystems and human livelihoods (Brisbane Declaration, 2007; Acreman & Ferguson, 2010). Environmental flow regimes were initially driven by the concern for the survival of salmonid populations, which are highly sensitive to physical habitat alterations. However, environmental flows are also required to support all aquatic biota including, migratory and non-migratory fish, birds, invertebrates, aquatic vegetation and mammals.

The most common approach to determine minimum flows for run-of-river hydropower schemes, which tend, incorrectly, to be considered synonymous with environmental flows, is to allocate a fixed minimum amount to the depleted reach based on the flow duration curve (a “Q” value) or a percentage of the mean natural flow (Figure 2.9). The Q value is the percentage of time a particular flow amount is exceeded, e.g. a Q_{95} is the flow exceeded for 95% of the time. A residual flow through bypass channels equivalent to Q_{95} is often described as an appropriate environmental flow for run-of-river hydropower schemes, with the value being equivalent to a natural low flow event in some rivers (A. Butterworth, pers. comm.). However, setting such broad environmental flows is difficult as each river may have a different flow regime, geomorphological structure, and subsequently different fish community structure and riverine processes. Additionally, each proposed run-of-river hydropower scheme may have different economic operational requirements that will influence the amount of water required for abstraction. There must be a maximum abstraction limit for hydropower however. This ensures that when abstraction is taking place, particularly in valuable stretches such as salmonid spawning areas, essential high winter flows will, for example, pass down the depleted reach to clean gravels, a function fundamental to ensure their reproductive success; i.e. flow variability is maintained. It is unknown however whether the reduced flows are sufficient to clean the gravels, as although there is an abstraction limit, high spate flows in winter are still abstracted.

Finally the flows along each river system must be sufficient to ensure no net deterioration of ecological status under the terms of the WFD. As a consequence, UK environmental flow indicators were revised through an expert consensus workshop (Acreman *et al*., 2006). The work involved derivation of river typologies and subsequently resulted in the definition of restrictive flow management recommendations for different types of small-scale run-of-river hydropower schemes (Tables 2.3 & 2.4). These provide default minimum flows relating to river type based on Q_{95} and Q_{mean} ratios (EA Good Practice Guidelines; EA, 2009a) or Q_{90} or Q_{95} (SEPA Guidance on Run-of-River Hydropower Schemes; SEPA, 2010a). The latter also offers guidance on maintaining natural flow variability and protecting high flows and flows for upstream movement and spawning of fish (Table 2.4).

Table 2.3 Maximum hydropower flows (Max) and Hands-Off Flows (HOF) according to river type (Q_{95}:Q_{mean} ratio) (EA, 2009a).

Depleted Reach		Q_{95} : Q_{mean} ratio			
		Flashy river <0.1 Fish migration issues	Flashy river <0.1 No fish migration issues	0.1 – 0.2	High baseflow >0.2
Weir only	Max	Q mean	Q mean	Q mean	Q mean
	HOF	Q95	Q95	Q95	Q95
Up to 200 m	Max	Q 40	Q mean	Q mean	Q mean
	HOF	Q 90	Q 90	Q 95	Q 95
>200m	Max	Q 40	Q mean	Q mean	Q mean
	HOF	Q 85	Q 85	Q 90	Q 95

Table 2.4 Restrictive flow requirements for run-of-river hydropower schemes in Scotland (SEPA, 2010a).

Purpose	Mitigation requirement
Protection of low flows to avoid wetted width of channel being reduced significantly	No abstraction of flows at or below a hands-off flow equivalent of Q_{90} or Q_{95} dependent on site specific factors (see (in SEPA document))
Protection of flow variability to avoid extended periods of low flow downstream of intake	No extended period during which the flow downstream of the intake is at, or below, the hands off flow • Flow downstream increases in proportion to flow upstream rising to Q_{80} when upstream flow would be Q_{30}; or • scheme shuts down for a fixed period at an agreed frequency designed to ensure flow higher than the hands-off flow occurs with equivalent frequency
Protection of high flows to maintain natural disturbance regimes and sediment, transport and erosion processes.	Maximum abstraction not to exceed 1.3-1.5 times the mean daily flow (usually around Q_{30}) dependent on characteristics of the scheme.
Protection of flows that trigger upstream movement and enable fish to pass natural and artificial obstructions and provide sufficient flows for fish to progress upstream.	Good status flows are maintained across the relevant flow range (i.e. flows up to Q_{10}) during periods of migration and spawning.

The SNIFFER WFD 48 expert panel also provided a summary of the key river typology requirements related to WFD ecological status (Table 2.5) and key fish community requirements (Table 2.6).

Table 2.5 UKTAG flow standards for rivers of high status.

Water resources standards for rivers of high status					
Types		Flows > QN_{95}		Flows < QN_{95}	
		(allowed per cent change from the natural flow)			
All Types		Up to 10		Up to 5	
Water resources standards for rivers and good status					
Types	Season	Flow > QN_{60}	Flow > QN_{70}	Flow > QN_{95}	Flow < QN_{95}
		(allowed per cent change from the natural flow)			
A1	April – Oct	30	25	20	15
	Nov– March	35	30	25	20
A2 (d/s), B1, B2, C1, D1	April – Oct	25	20	15	10
	Nov– March	30	25	20	15
A2(headwaters),C2, D2	April – Oct	20	15	10	7.5
	Nov– March	25	20	15	10
Salmonid spawning and nursery areas (Not chalk rivers)	April – Oct	25	20	15	10
	Nov - March	20	15	Flow> QN_{80} 10	Flow <QN_{80} 7.5

Type A1 – Clay and / or chalk; low altitude; low slope, Type A2 – Eutrophic; silt gravel bed, Type B1 – Hard sandstone and limestone, low/medium altitude; low/medium slope, Type B2 – mesotrophic with gravel boulder or pebble-cobble bed, Type C1 – Non-calcareous shales, hard limestone and sandstone, medium altitude, medium slope, Type C2 – oligomeso-trophic with pebble, cobble and/or boulder bed, Type D1 – Granites and other hard rocks; low and high altitudes, gentle to steep slopes, ultra-oligo, Type D2 – Oligo-trophic with cobble, boulder, bedrock and / or pebble bed.

Table 2.6 Summary of restrictive management requirements regarding threshold abstractions for key fish communities (after Acreman *et al.,* 2006).

Fish community type	Time of restriction	Threshold abstraction level
Chalk river fish communities (Trout and grayling)	All year	20% any flow on the day < Q_{95} 10% < Q_{99} 5%
Eurytopic/limnophylic fish communities (Roach, bream, tench, pike, bleak)	July-April	50% at medium high flows Hands-off at Q_{98}
	May-June	20% at medium high flows Hands-off at Q_{98}
Rheophilic cyprinid fish communities (Dace and chub)	February-June	50% Annual Q_{90} hands-off
	July-January	50% < Q_{90} 25% < Q_{95} 20% Q_{99} hands-off
Adult salmonids (other than Chalk rivers)	All year	50% of flow above Q_{95} Hands-off at Q_{95}
Salmonid communities (spawning and nursery areas)	June-September	20% abstraction of flow above Q_{95} Hands-off at Q_{95}
	October-May	Hands-off at Q_{80}

Whilst there remains debate about the precision and applicability of these minimum flow standards, they are based on the best available information at the time of derivation and are aimed at providing protection to different fish community types typically found in UK rivers.

2.3.4 Turbine types

Hydropower turbines can generally be classified into two categories: reaction and impulse (Figure 2.12). Reaction turbines include the water wheel, Tyson, Kaplan (propeller, bulb, straflo and tube), Francis and Archimedean screw. In these types, water changes pressure as it moves through the turbine, releasing energy. By contrast, impulse turbines change the direction of flow of a high-velocity water jet; consequently they extract energy from the “impulse” of moving water. Impulse turbines are most often used in high-head schemes. Turbines within this category include the Pelton, Turgo and Michell-Banki (also commonly referred to as the crossflow or the ossberger) (Paish, 2002). It is common for most types of turbines to utilise the principles of both impulse and reaction turbines, but there are exceptions such as the Pelton, which exclusively uses the impulse concept. Crossflow turbines are designed as an impulse machine but will maintain some efficiency through reaction in low-head applications like a traditional water wheel. The crossflow turbine was developed to accommodate larger water flows and lower heads than the Pelton (Nelson, 2011).

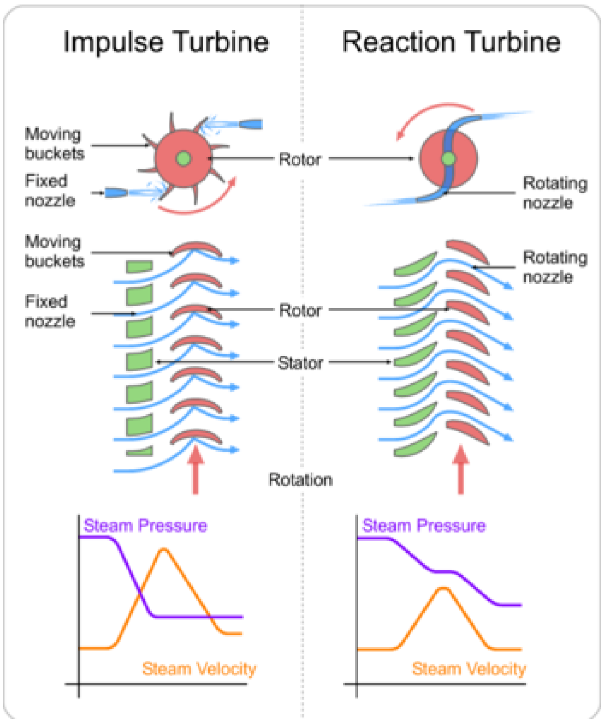

Figure 2.12 Principles of impulse and reaction turbines (NWE, 2011).

2.3.4.1 Reaction turbines

Reaction turbines use the potential energy of flowing water to exert a hydrodynamic force onto a blade, causing it to rotate. This is the commonest type of turbine for low-head installations, and Francis and propeller turbines, such as Kaplan, are examples. The Francis turbine is the most common water turbine in use today, and can operate in a head range anywhere between 10 m and several hundred metres. Kaplan turbines are propeller-type water turbines that have adjustable blades and consequently are considered a step up from the Francis turbines. The adjustable blades allow this turbine to work efficiently in low-head schemes, which the Francis turbine cannot achieve. Tyson turbines are inserted directly into flowing water, extracting power from the flow of water. It requires no local engineering and is easily moved between locations. Water wheels were more commonly known to power mills; however, they are now being used in the production of small-scale hydropower. The Archimedean screw turbine operates with a slow rotational speed, very low shear forces and no pressure changes (Kibel, 2008; Kibel *et al.,* 2009).

2.3.4.2 Impulse turbines

Impulse turbines use the kinetic energy of the flow that goes from the nozzle of a pressure pipeline to the turbine blades, making the turbine wheel (runner) rotate. Pelton turbines are frequently used on small high-head schemes. Although many impulse turbines existed prior to the development of the Pelton turbine, this is now considered to be one of the most efficient. This is because water leaving older impulse-type turbines would have relatively high speed, whereas the Pelton was designed to ensure the water leaving the wheel had very little speed, providing a very efficient turbine extracting almost all of the water's energy. The Turgo turbine is a modification of the Pelton and generally used in "medium"-head schemes. This design is improved by the runner being less expensive and it can handle a much greater flow than that of the Pelton. Crossflow turbines are well suited to low-head schemes with high flow. Their advantages are low cost, simple construction and ease of maintenance

2.3.5 Turbine design

Previous studies suggest hydro-electric turbines can be the cause of considerable mortality; estimated 20% for Francis turbines, 12% for Kaplan turbines and 9% for bulb turbines (Odeh, 1999b; Pavlov *et al.,* 2002). Larinier (2008) found considerable variability in the mortality rate for juvenile salmonids passing through Kaplan (5 and 20%) and Francis (under 5% to over 90%) turbines thus highlighting the extreme importance of turbine design. Turbine design, such as the type of head, number of blades, and rotation speed, is an important factor with respect to survival of fish going through turbines. Horizontal axis, adjustable (bulb) turbines cause the lowest mortality, followed by horizontal axis adjustable (Kaplan) turbines. Vertical axis, fixed (Francis) turbines and impulse turbines (Pelton) have the lowest survival ratios. Consequently, one area of action is to improve the design of turbines to minimise injury and mortality of fish passing through the units.

Odeh (1999b) reported several mechanisms that could make Kaplan turbines both fish and environmentally 'friendly' and simultaneously increase efficiency of the turbine. Improvements include: operating the turbine at high efficiency with reduced back roll and no cavitations; removing the gaps within a turbine system; eliminate gaps between the blades and the hub and discharge ring, preventing grinding; correct placing of wicket gates and stay vanes; use of lubricating fluids and greases that are biodegradable - this will ensure the surfaces of stay vanes, wicket gates and draft tube cone are smooth; welds on parts of the turbine system can be kept smooth to reduce abrasion to fish; installing sounding devices to reflect when trash racks need cleaning as flow disturbance is minimised when trash racks are clean and will allow fish to enter the upper part of the intake rather than the lower portion, reducing blade tip strike.

Similarly, Francis turbines can be modified to become fish 'friendly' with some factors resembling those of the Kaplan turbine. Reducing the number of blades (but increasing their length) reduces the probability of strike but increases the size of flow passage and using a thicker blade edge, reducing the overhang of wicket gates, enhancing the wicket gate to runner distance, ensuring the stay vanes and wicket gates are aligned, operating the turbine with adjustable speeds and minimising pressure changes by providing fish with a passage route that will reduce fluctuating pressure changes. Francis turbines could also be aerated, increasing dissolved oxygen levels in the turbine discharge (Odeh, 1999b).

The Voith-Siemens Minimum Gap Runner (VLH) turbine (www.vlh-turbine.com) also claims a better level of fish protection than conventional turbines at low-head sites. Trials by injecting eels directly into the runner suggests an average survival rate of 95%, although those individuals injected into the periphery of the turbine only showed an 84% survival rate (LeClerc, 2008). The test suggests that the combination of greater efficiency and higher capacity of the VLH means that it could generate about 14% more electricity than a Kaplan turbine installed at the same site.

Despite these modifications, there is pressure to develop more 'environmentally friendly' turbines to enable use of a reliable source of renewable energy whilst simultaneously maintaining a healthy ecosystem (Odeh, 1999b). A number of 'fish friendly' turbines have also been developed in recent years (e.g. Cada, 1998; Hecker &

Cook, 2005), and one of the most promising for low-head sites appears to be the hydrodynamic, or Archimedean screw. These offer a greater degree of protection to downstream migrating fish (Spah, 2001; Merkx & Vriese, 2007). The Archimedean screw turbine operates with a slow rotational speed, very low shear forces and no pressure changes (Kibel, 2008; Kibel *et al.,* 2009). Spah (2001) and Kibel (2007, 2008) found that provided certain precautions were taken to buffer the leading edge of Archimedean screw systems to make sure there are no “pinch” points, even eels, large trout and salmon kelts passed safely downstream across a wide range of sizes and operating speeds.

Tests undertaken with different leading edge protection (Table 2.7; Figure 2.13) emphasise the importance of design, confirming damage to fish can be severely reduced with the correct design.

Table 2.7 Damage sustained to fish by different leading edge adaptations of Archimedean screw system.

Leading edge profile	Turbine speed $m\ s^{-1}$	Number of fish	Number struck	Damage sustained
Unprotected steel edge	4.5	10	1	Significant haematoma
Hard rubber profile	4.5	10	3	Minor haematoma
Compressible bumper	4.5	10	2	No damage

Steel edge (significant damage) | Hard rubber (some damage) | Compressible bumper (no damage)

Figure 2.13 Damage to fish caused by different leading edges of Archimedean screw turbines (Source: Fishtek Consulting, 2009).

Kibel *et al.* (2009) assessed three leading edge profiles (8-mm unprotected blade, 20-mm hard rubber and a compressible bumper with 35 mm of compression) across a range of speeds and fish sizes. Based on their findings, it was recommended that compressible bumpers should be fitted to screw turbines as this offered effective mitigation in comparison to the other leading edge profiles. It should be noted however, that these results are not considered to be based on sound science as some species used in trials were not live, thus results should be treated with caution and indicative only.

Ultimately, however for most turbines, the only effective way to improve survival is to direct the fish away from the intakes using screens or louvers and then, in both low and high-head schemes, through systems that bypass the turbines. Directing more of the flow across spillways or through downstream bypass facilities may also help reduce mortality. These measures may, however, reduce the generating capacity of the plant, particularly in times of low flow. For high-head schemes, screening is the only possible solution, although there is usually no need to consider bypass facilities because the intake structures tend to be above an impassable barrier and fish species tend to be isolated and resident.

2.3.6 Intake and outfall design

Potential impacts associated with erosion, scour and sediment deposition can arise as a result of the intake and outfall locations and design. There are several different designs regarding the intakes and outfalls of run-off-river hydropower, each of which should be carefully chosen according to the scheme in question. The design of each feature will vary depending on the surrounding environment and criteria of the scheme, including abstraction properties, site conditions and type of discharge. There are five types of intakes associated with high-head-schemes:

- Submerged – the intake is submerged under water on the bottom of the river bed
- Bed intake – the intake covers the width of the channel and is buried in the river bed
- Bank-side with no in-stream structure – the intake is built into the bank with no flow detection or fore bay structures
- Bank-side with in-stream structure – the intake is built in to the bank with a flow detection structure or fore bay
- Bank-side with weir – the intake is built into the bank drawing water from a storage area behind a weir

Many of the intakes used within Scottish schemes follow the design based on a bank-side with weir.

In addition there are two types of intake associated with low-head schemes:

- Off-take on weir
- Diversion with weir – this is typical of low-head schemes that make use of diversion channels such as leats (see Figure 2.6).

There are four types of outfall:

- Bank side
- Submerged
- Partially submerged
- Set back, with one or a series of inlets

A stable bank with no undercutting or erosion would be an ideal location for an outfall.

The location and design of the intakes and outfalls can go a long way to minimising problems with fish entrainment into turbines or disruption of upstream migration of fish, but there will usually remain a need to further prevent any likelihood of fish being entrained or diverted away from the optimal migration route, and appropriate screening should also be installed.

2.3.7 Screening

Screening is vital, especially in high-head run-of-river schemes to prevent smolts and juveniles from entering the pipeline. In the context of low-head schemes there are two areas where screening may be needed: 1) if mortality of fish during passage through turbines is high, then a bypass or protection system is needed (Godinho & Kynard, 2009); and 2) where discharge at the tailrace of the turbine attracts upstream migrants. Damage to fish passing through turbines is a major cause for concern of newly proposed hydropower schemes. To mitigate this problem, a number of solutions are available (see EA, 2009a and SEPA, 2010a). These include the placement of intake screens and other bypass systems, including surface collectors and barges, which steer or transport fish away from the intakes. The most common and effective measure to protect fish from entrainment (Clay, 1995; Kynard, 2004) is screening of the turbine water intake, especially in circumstances where a downstream fish pass is not provided or perhaps not necessary. There is also a need to prevent fish that may be attracted to a discharge flow from entering the turbine discharge or being distracted away from the main natural flow (Vovk-Korz *et al.,* 2008). Specific consideration should be given to the following additional aspects at hydropower sites (EA, 2005):

- the risk of fish injuries or mortalities in the turbines (via both intakes and outfalls);
- possible delays in fish migration and increased predation risk when water is

diverted through long head- and tailrace systems;

- possible losses at bywash outfalls where the increased concentration of diverted fish may attract predators, especially if fish are disorientated

Solomon (1992), Turnpenny *et al.* (1998) and O'Keeffe & Turnpenny (2005) provide comprehensive accounts of appropriate screening. Screens need to be designed for the type of turbine and the expected species and size of fish, and should be angled to direct fish towards a suitable channel or sized to reduce the approach velocity to the sustainable swimming speed of the fish. Currently, there are two categories of screening: behavioural and physical.

Behavioural screens utilise the avoidance reaction of fish to a repellent stimulus such as strobe lighting or sound pulses. Many types of behavioural barriers have been tested, but so far few have worked consistently well enough to be adopted in operational hydropower schemes. Visual, auditory, electrical and hydrodynamic stimuli have been used at experimental barriers, but none have proved effective because of their specificity (to species and size), low reliability and susceptibility to local conditions. Maes *et al.* (2004) investigated an acoustic deterrent system producing 20-600 Hz to deter estuarine fishes away from the water inlet of a power station. Total fish impingement decreased by 60% during sound emission, but the avoidance response varied amongst species. The effectiveness of the acoustic system appears to depend on whether the fish had an accessory structure that would increase hearing abilities; those fish without swim-bladders generally showed no or little response, whereas the intake rates were significantly reduced in species with swim-bladders. Furthermore, fish become accustomed to the screen stimuli and they become no longer effective.

Physical screens actively prevent fish from being drawn into the turbines and include mesh screens, louvered bars and angled bar racks above the intakes, and vertical curtains suspended in the forebay. Most of these techniques have been successfully deployed in temperate rivers in the northern hemisphere where they are used primarily to reroute migrating salmon. These methods only work for large, comparatively strongly swimming fish; they are not suitable for passively drifting eggs or larvae, or for small fish (Benstead *et al.,* 1999). The positioning of the screens is critical to avoid areas of high current velocity that enable the fish to actively avoid entrainment onto the screen (Amaral *et al.,* 2003). The risk of entrainment on screens varies according to fish species and size; smaller eels for example are able to pass through the bars and are inevitably damaged or killed; depending on the type of turbine. The screens also have to be continuously maintained and cleaned to be practical.

To prevent this, smaller-mesh screens with lower through-screen velocities (e.g. Figure 2.3) as well as effective alternative passage routes can be provided (Boubee & Williams, 2006). The bars or meshes will differ in spacing according to the body shape and size of the fish being protected (Table 2.8).

Table 2.8 Default screen characteristics for different turbine intakes (after EA, 2009a).

Turbine type	**Default screen**
Archimedean screw Water wheel	Trash only, but bywash required
Low-head Francis Kaplan >1.5 m dia	10- or 12.5-mm bar
Spiral Francis Kaplan <1.5 m dia Crossflow	6-mm bar
Impulse (Pelton)	3-mm drop through

In Scotland, a rectangular mesh size of 12.5-mm vertical x 25-mm horizontal gap is used, and generally is accepted by the District Salmon Fisheries Boards (Turnpenny *et al.,* 1998). SEPA (2010a) guidance, however, states that developers are expected to install screen intakes with the default maximum gap size of 10 mm; coanda screens are preferred where possible. The coanda screen surface is comprised of wedge-wire, contoured to form an ogee-shape curved to a 3 m radius. Bars run from side to side across the width of the weir with spacing between the wedge wire bars designed to be small enough to exclude all fish; even larvae and juveniles (O'Keeffe & Turnpenny, 2005). Flexibility is still provided, for instance where the use of coanda screens are impracticable, other types of screens may be used. Larger gap sizes of screens may also be used but only when certain criteria relating to the characteristics of the fish populations at a site apply.

Although in Scotland, coanda screens are generally recommended, a further possible option, especially for high-head hydropower schemes, is the wedge wire cylinder screen, e.g. Johnson screens. Wedge wire cylinder screens provide relatively low-maintenance passive screening systems (Solomon, 1992) and are preferable for smaller abstractions such as those on high-head hydropower schemes (Turnpenny *et al.,* 1998). Generally a slot width of approximately 3 mm is used; this ensures fish of almost any size, eggs and larvae are successfully prevented from entering. Unfortunately, these screens are considered to be uneconomic/impractical where abstraction flows are more than a few m^3/s (Turnpenny, 1989). Spillway screens have been used in the USA for more than a decade, but are not as common in England. They are, however, common in Scotland and SEPA's preferred screening method of choice. A spillway screen relies on the principle that a grid replaces a portion of the downstream face of a weir, water that falls through the grid then enters a channel underneath, whilst fish that are larger than the screen openings are carried by the surplus flow over the grid surface to the downstream side of the weir (Turnpenny *et al.,* 1998).

As indicated previously, upstream migrating fish may be diverted or delayed if they are attracted into tailrace channels. They must therefore be prevented from doing so. Physical barriers are the best option to overcome this problem for both salmonids and non-salmonid fish. Wedge wire, square or oblong metal bars with 40 mm spacing for salmon and 30 mm for sea trout, are recommended; round or oval mesh are not recommended because they are more likely to gill fish. Unless smaller species are absent, recent SEPA guidance specifies a maximum of 20 mm spacing. The screen should be placed close to the edge of the river bank at the point of return of the turbine discharge to the water body. Screens should be positioned in such a way that downstream migrating fish that successfully pass through the turbines are allowed to continue their migration. Although proposed in the Environment Agency Good Practice Guidelines Hydropower Handbook (EA, 2009a), the use of electric screens are not recommended; fish become tolerant to the electric field and easily pass through them making the screen ineffective (Cowx & Lamarque, 1990). Electric screens also impose considerable health and safety problems that are not easily overcome.

It is important that outfall screening remains effective even when the outfalls are not discharging. Otherwise, fish may enter them during quiet periods, subsequently risking becoming trapped or injured (EA, 2005).

2.3.8 Provision for sediment transport

As indicated in Section 2.2.4, run-of-river schemes have the potential to cause accumulation of sediment upstream of the impounding structure, thus disrupting sediment supply to river reaches downstream. Where this is the case, measures should be taken to re-supply those reaches with sediment that occurs upstream of the intake structure. The SEPA Guidance for Run-of-River Hydropower Schemes (SEPA, 2010a) offers on management sediment under such circumstances. It is recommended that natural sediments are reintroduced to a suitable location downstream that is as close as possible to the intake structure. The accumulations can be returned by:

- designing the intake structure such that high flows move sediments over the impounding structure and into the river downstream;
- operating scour valves (although this is not considered realistic at the impoundments associated with small-scale schemes); or

- excavating, transporting and reintroducing the sediments.

SEPA (2010a) also recommend the sediment is returned during periods of high flow (to aid redistribution of the sediments), at locations that will not impede the free passage of migratory fishes and during periods that will not interfere with spawning or between spawning and the emergence of juvenile fishes.

2.4 Review of UK regulations

Dependent upon the size of the scheme in Scotland, different authorities must be contacted to grant development consents (Table 2.9). While Scotland, England & Wales and Ireland have all released guidance manuals for run-of-river hydropower schemes there are differences in certain aspects of the design criteria between the regions (Table 2.10) and also regulatory differences between administrations. For example, while SEPA set a hands-off flow ranging between Q_{90} to Q_{95} the hands-off flow set by the EA can range between Q_{85} to Q_{95}. Differences in the mesh size of the tailrace screens are also apparent. The EA set standards as 40 mm and 30 mm spacing for salmon and sea trout respectively, while SEPA sets the default as <20 mm.

Table 2.9 Development consents in Scotland.

Size (installed capacity)	Consent	Determining authority	E.I.A
>1 MW	Section 36, Electricity Act 1989	Scottish Ministers	Yes
>500 kw to ≤1MW	Town & Country Planning Act 1997	Local Authority	Yes
≤500 kW	Town & Country Planning Act 1997	Local Authority	If in sensitive area

2.4.1 CAR Licence

The Water Environment and Water Services (Scotland) Act 2003 is the enabling act for the European WFD, which introduced a new integrated approach to the protection, improvement and sustainable use of the water environment. The Water Environment (Controlled Activities) (Scotland) Regulations 2011 (CAR) introduced controls on previously unregulated activities, including water abstractions and impoundments, which is of significant relevance to hydropower developments. As such for each hydropower scheme that is proposed in Scotland, developers must be granted a CAR licence from SEPA. This provides authorisation for abstractions, impounding works including weirs and dams, and any other engineering works associated with the development of the scheme. Table 2.11 outlines the key aspects of how regulation is carried out in the different parts of the UK.

Table 2.10 Comparison of the design criteria from the NIEA, SEPA and EA guidelines for run-of-river hydropower schemes.

	Northern Ireland Environment Agency	**Scottish Environment Protection Agency**	**Environment Agency**
Hands-off flow The minimum residual flow at any time down the stretch of river between the intake and the outfall, when generation is taking place	Default Q_{95} Q_{90} – For rivers designated as salmonid under the Freshwater Fish Directive Q_{80} under the following circumstances: All rivers designated as an SAC, Tributaries of SAC rivers where the conservation features includes salmonids and their habitat	Default Q_{95}* Q_{90}* under following circumstances: Sites with populations of salmon or sea trout. Sites designated for the conservation of aquatic plants or animals. Sites with catchment area upstream of the tailrace < 10 km^2. Sites where the wetted width is significantly reduced at flows below Q_{90}	Q_{85}-Q_{95} – see Table 2.3
Protection of flows	Stopping abstraction for an agreed fixed period. Provision of seasonally variable flows – building blocks. Releasing of freshets through short period shut downs	Good status flows – flows up to Q_{10} maintained during periods of migration and spawning. Scheme shuts down for fixed periods at an agreed frequency	Adequate flow in depleted reach during migration seasons. Where there is no fish pass, adequate residual flow over the weir during migration seasons
Max abstraction	Q_{mean} – (Q_{30}-Q_{40}) 1.5 times the Qmean considered when: Impounding structure associated with project does not impede sediment transport. Impacted depleted stretch has low habitat value. No fish population.	1.3-1.5 Q_{mean} (1.3 or 1.5 times the average daily flow depending on site characteristics)	Q_{mean} – Q_{40} Q_{40} if fish migration issues Q_{mean} if no fish migration issues
Screens compulsory	Applicants must ensure that all abstraction intakes are appropriately screened and the discharge point must be appropriately screened. All developments must have the screening at the point of abstraction from the river, however, where this is not feasible for technical reasons, then a smolt by-pass should be installed	No For downstream passage of fish if scheme uses Archimedean screw, screening is not necessary providing suitable protection of the leading edge such as compressible silicone extrusion. In this instance the tail race must also be unscreened	No If scheme uses Archimedean screw, screening is not necessary providing suitable protection of the leading edge such as rubber bumper or a compressible silicone extrusion. In this instance the tail race must also be unscreened

	Northern Ireland Environment Agency	Scottish Environment Protection Agency	Environment Agency
Intake screen mesh size	Minimum 51 mm	Default screen is a coanda screen, others (i.e. 10 mm) only acceptable if coanda is not feasible ≤ 10 mm Screen gap sizes >10 mm may be acceptable if: the proportion of salmon and sea trout smolts present that have a length < 11.5 cm is insignificant; no ecologically significant downstream movement of salmon or trout fry/parr, juvenile eel or lamprey occurs in the part of the river	Dependant on turbine type: Archimedean screw – trash screen 100 mm Kaplan & Francis (≥1.5 m^3/s) – 10 12.5 mm (dependent on region) Kaplan & Francis (≤1.5 m^3/s) – 10-12.5 mm (6 mm in summer when fry are present) Pelton & Impulse – coanda 3 mm – 6 mm if salmonid fry, under-yearling coarse fish and lamprey ammocoetes are absent
Tailrace screen mesh size	Minimum 25 cm	Default <20 mm Screen gap sizes > 20 mm may be acceptable if: The tail race flows will not attract upstream moving fish. Evidence is provided that adult brown trout do not move upstream past the proposed site of the tailrace to spawn	40 mm spacing for salmon 30 mm where sea trout are present
Screen type	Physical screen or electric barrier	Physical coanda or drop screens. Screens constructed from wedge wire, square or oblong metal bars. Round or oval bars should NOT be used	Physical screen or electric barrier screens constructed from wedge wire, square or oblong metal bars. Round or oval bars should NOT be used
Fish pass compulsory	On migratory salmonid rivers where there is currently no fish pass, then normally it is expected that an appropriate fish pass should be installed. On other rivers a fish pass may be necessary where it is considered that any reduction in fish passage may	Downstream passage is required at all sites, but upstream only where migratory fish (including brown trout) are likely to migrate past the intake.	Yes, if salmon, migratory (sea) trout are present or if there are objectives to rehabilitate them to the river. No, if fish are absent and will continue to be absent following achievement of

	Northern Ireland Environment Agency	Scottish Environment Protection Agency	Environment Agency
	cause deterioration in ecological status or that the absence of one is preventing achievement of Good Ecological Status		the objectives of the river basin management plan or, alternative provisions for fish passage
Licenses	Water abstraction and impoundment (licensing) regulations (Northern Ireland) (2006)	Water environment (Controlled activities) (Scotland) Regulations 2011 (CAR) Development consents (see Table 2.9)	Impoundment licence. Full or transfer licence. Flood defence consent. Fish pass approval
E.I.A		Yes if scheme is >500 kW or located in a sensitive area	Yes if scheme is >500 kW
Determination of licence application		4 months	
Regulations and Directives	EU Habitats Directive WFD EU Birds Directive Ramsar Convention The Foyle Fisheries Act (NI) 1952 The Fisheries Act (NI) 1966 The Fresh Water Fish Directive The Drainage (NI) Order 1973 The Renewables Directive The Renewable Energy Directive Single Energy Market Directive Renewables Obligation Order (NI) 2005 Energy Order (NI) 2003	EU Habitats Directive WFD EU Birds Directive Electricity Works (Environmental Impact Assessment) (Scotland) regulations 2000 Environmental Impact Assessment (Scotland) Regulations 1999 Salmon (Fish passes and Screens) (Scotland) Regulations 1994	EU Habitats Directive WFD EU Birds Directive Salmon and Freshwater Fisheries Act of 1975 Fish Pass approval Fish Screen regulations
Organisation involved	External consultation The Rivers Agency The Loughs Agency Department of Culture, Arts and Leisure (DCAL) Internal Consultations Natural Heritage Built Heritage The Freshwater Monitoring and Assessment Team The Industrial Consents Team Northern Ireland Water Regulation Team River Basin Planning Team	Local Planning Authority Scottish Government Scottish Natural Heritage District Salmon Fishery Board Marine Scotland Rivers and Trusts of Scotland	Countryside Council for Wales (CCW) Natural England (NE) Local Governments National Park Authorities Local Nature Reserves Authorities Local conservation officers

Table 2.11 Regulatory comparisons between administrations.

Region	Planning body - (contact)	Environmental regulatory body	Primary constraint on hydropower imposed by regulator on basis of :	Abstraction & impoundment controls	Flooding and drainage controls	Fish passage controls and enforcement	Advisers on fish passage	Environmentally Protected Areas	Heritage
England	Local government planning authority - Planning Portal (England and Wales)	Environment Agency (EA)	New impoundment abstraction	Transfer abstraction licence (EA) Impoundment licence (EA)	Land drainage consent (EA)	Fish pass approval (EA)	EA internal	Natural England	English Heritage
Wales	Local government planning authority - Planning Portal (England and Wales)	Environment Agency (EA)	New impoundment abstraction	Transfer abstraction licence (EA) Impoundment licence (EA)	Land drainage consent (EA)	Fish pass approval (EA)	EA internal	Countryside Council for Wales	Cadw
Northern Ireland	Northern Ireland Planning Service - Northern Ireland Planning service	Northern Ireland Environment Agency (NIEA)	New impoundment abstraction	Abstraction licence (NIEA) Impoundment licence (NIEA)	Northern Ireland - Rivers Agency approval	Loughs Agency Fisheries Conservancy Board for Northern Ireland	Loughs Agency Northern Ireland - Department of Culture, Arts and Leisure	NIEA Protected Areas	NIEA Protected Areas
Scotland	Scottish Government (>50 MW, S36 Electricity Act consent) Local government planning authority (< 1 MW) – Scottish Government: Planning authorities	Scottish Environment Protection Agency (SEPA)	Scheme generation capacity then new impoundment abstraction	Controlled Activities Regulation (CAR) authorisation	CAR authorisation	CAR authorisation Fish passage and screen regulations	SEPA internal, Marine Scotland, District Salmon Fishery Boards	Scottish Natural Heritage	Historic Scotland

CHAPTER THREE

3. GENERAL MATERIALS AND METHODOLOGY

3.1 Study Area

The River Tay is the longest river in Scotland, with a length of 193 km stretching from the northern slopes of Ben Lui to the Firth of Tay near Dundee. The headwaters of the River Tay are formed by the tributary rivers of the Dochart, Lochay and Lyon and much of the area's native woodlands and wider biodiversity is associated with these tributaries (Tay Western Catchments Project, 2010). The Tay river system is the most extensive in Scotland, draining a catchment in excess of 5000 km^2. Furthermore, approximately 10,000 ha of the River Tay is designated as one of the 17 Scottish Special Areas of Conservation (SAC) for Atlantic Salmon. Sea lamprey, river lamprey and brook lamprey are present in the River Tay and also contribute to the SAC designation as a qualifying feature. Whilst not a designated feature, the River Tay hosts one of the largest known freshwater pearl mussel populations in the UK (and therefore the world). SACs are designated under the European Directive commonly referred to as the Habitats Directive; SACs form the Natura 2000 network of sites. SAC designation is recognition that some or all of the wildlife and habitats are particularly valued in a European context.

The River Tay is one of the freshwater fish protected areas, in Scotland, defined as areas designated for the protection of economically viable freshwater fisheries, specifically salmonids. Due to the high-quality Atlantic salmon populations the Tay supports, many concerns and issues surround the development of run-of-river hydropower schemes within the Tay catchment, causing open debates between parties involved. Rod catch returns consistently convey the Tay to be one of the top three salmon rivers in Scotland; as such the communities that live along the Tay have strong economic, environmental and historical ties with it. As the river provides some of the best salmon fishing this subsequently creates benefits to all Tay economies through the highly lucrative influx of fishermen from all over the UK and indeed Europe; additionally the river also caters extremely well for local anglers fishing for salmon and a range of other species including brown trout and grayling (Tay Western Catchments Project, 2010). Although grayling are non-native to Scotland, they are the source of considerable angling interest. The high quality salmon populations found in the River Tay are of great economic importance; salmonids are known for their intrinsic recreational and commercial value (Scott & Crossman, 1998) and are estimated to bring in £50-100 million per year to the Scottish economy from recreational fisheries

(Scottish Office, 1997). In 2004 game and coarse anglers contributed £113 million to the Scottish economy with salmon and sea trout anglers accounting for over 65% (£73 million) of this total (Radford *et al.,* 2004). Salmon and freshwater angling are crucial to the economic well-being of the area and as fish are an important indicator species of an ecosystem's health, the status of fish populations therefore provides a useful barometer of biodiversity and water quality. The River Tay supports the entire range of salmon life history types (see Section 1.3.1.1).

As part of WFD it is an objective to improve the remaining failing water bodies to good ecological status or potential, either by 2015 or over the first three river basin planning cycles. In the Tay area, SEPA aim to have 338 water bodies (95%) at good or high ecological status by 2027. To achieve this, those currently at good or high ecological status will be protected from deterioration, while action will be taken to enhance and restore others. Protecting the status of a water body does not just mean preventing deterioration of their overall status. The overall status depends on the condition of several different elements including the plant community, fish populations and water quality. Currently, within the Tay area 170 (48%) water bodies are at good or high ecological status and 185 (52%) are at less than good status. For 17 water bodies in the Tay region, however, SEPA believe that good ecological status cannot be achieved, even by 2027. For these water bodies, a less stringent objective than good ecological status has been set. Three of these water bodies include Kinnaird Burn, Keltney Burn and Innerhadden Burn, each with a current status of bad, remaining in bad status by 2027 (SEPA, 2011). The reason for these less stringent objectives is that abstraction within these water bodies is causing a change to natural flow conditions. These three water bodies are amongst the ten rivers surveyed within this project, thus a run-of-river hydropower scheme is operational on each river, which could potentially compromise the achievement of good status.

The ten run-of-river hydropower schemes studied in this investigation (numbers in brackets indicate the location of each scheme; Figure 3.1) were:

- Kinnaird Burn (1)
- Keltney Burn (2)
- Rottal Burn (3)
- Innerhadden Burn (4)
- Inverhaggernie Burn (5)
- River Callop (6)
- Ardvorlich Burn (7)
- Douglas Water (8)
- Camserney Burn (9)
- Allt Gleann Da-Eig (10)

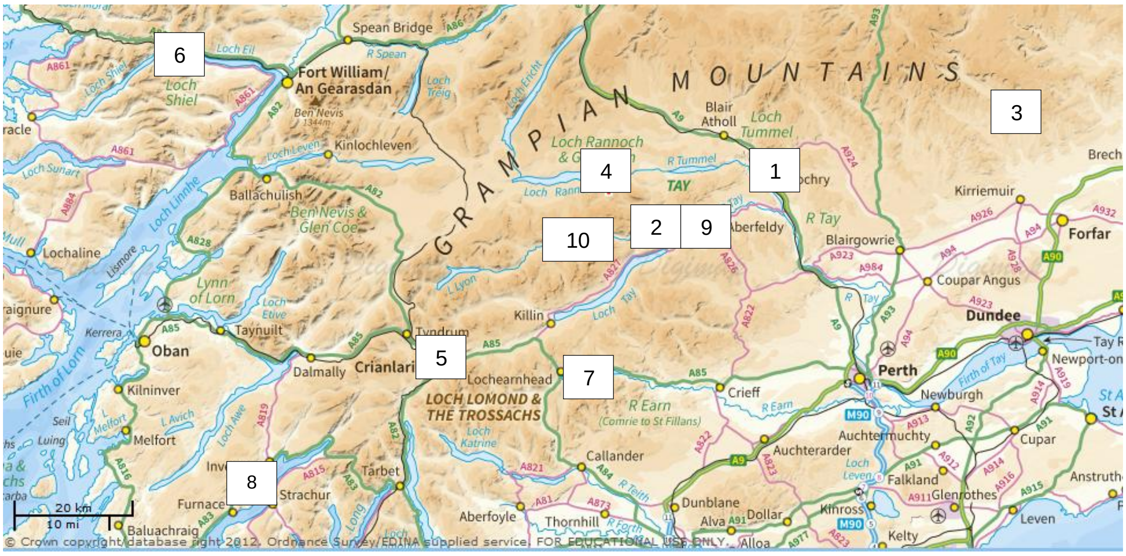

Figure 3.1 Location of the ten run-of-river hydropower schemes studied in this investigation. 1 = Kinnaird Burn, 2 = Keltney Burn, 3 = Rottal Burn, 4 = Inverhaggernie Burn, 5 = Innerhadden Burn, 6 = River Callop, 7 = Ardvorlich Burn, 8 = Douglas Water, 9 = Camserney Burn, 10 = Allt Gleann Da-Eig.

Each of the run-of-river hydropower schemes listed above are located within the Tay catchment with the exception of Rottal Burn and the River Callop. Rottal Burn is a tributary of the River South Esk, located in the South Esk catchment. The River South Esk is designated as a SAC, the main reason being the presence of high quality Atlantic salmon populations and freshwater pearl mussel, but most importantly is also recognised as one of Scotland's sea trout rivers. The River Callop flows into Loch Shiel located in the Shiel catchment, also designated as a SAC; providing important nursery grounds for fish populations including Atlantic salmon, brown trout and European eel.

Ardvorlich Burn and Douglas Water hydropower schemes were surveyed by Morgan Fisheries Consultants (MFC) in 2008 and 2002 respectively and SEPA in 2011. The River Callop was surveyed by MFC and Lochaber Fisheries Trust (LFT) in 2006 and 2010 respectively and by the author in other years; data underwent the same analysis as data collected by HIFI (where the author carried out her study) where the sampling regime allowed. The remaining seven schemes were all surveyed by HIFI (Table 3.1).

Table 3.1 x represents the years sampling frequency of the ten hydropower schemes between 2002 and 2011 by HIFI[1]; MFC[2]; LFT[3] and SEPA[4].

	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Kinnaird Burn				x[1]	x[1]	x[1]	x[1]	x[1]	x[1]	x[1]
Keltney Burn				x[1]	x[1]	x[1]	x[1]	x[1]	x[1]	x[1]
Rottal Burn					x[1]	x[1]		x[1]	x[1]	x[1]
Innerhadden Burn				x[1]	x[1]	x[1]	x[1]		x[1]	x[1]
Inverhaggernie Burn					x[1]	x[1]	x[1]		x[1]	x[1]
River Callop					x[2]				x[1], x[3]	x[1]
Ardvorlich Burn							x[2]			x[4]
Douglas Water	x[2]									x[4]
Camserney Burn			x[1]	x[1]	x[1]	x[1]	x[1]	x[1]	x[1]	x[1]
Allt Gleann Da-Eig						x[1]	x[1]		x[1]	x[1]

As the sites on the River Callop, Ardvorlich Burn and Douglas Water were chosen (and surveyed previously) by MFC, LFT or SEPA the site codes were adopted in their original format for inclusion in this study.

It should be noted that the last two schemes, Camserney Burn and Allt Gleann Da-Eig are included within this investigation even though there are potential limitations regarding impact assessment. Studies at Camserney Burn were limited only to post operational fisheries data, but the scheme was included as it has been monitored annually since 2004 providing a long-term fisheries population dataset with the scheme operational; additionally the river has control sites outside the abstracted reach for comparison with sites within the abstracted reach. The Allt Gleann Da-Eig scheme was due for commissioning in spring 2011 but a number of construction delays meant the scheme did not become operational until October 2011 following the fisheries surveys; therefore data collected for this scheme are used as an example to assess any potential impacts of construction of the scheme.

In the context of this study, ‘population’ refers to the fish stock in the reach of river or stream immediately under the influence of the hydropower scheme, i.e. section between the intake and outfall, but also including the effect of any barrier and impoundment that may fall outside this reach.

3.2 Survey site locations

3.2.1 Kinnaird Burn

The Kinnaird Burn run-of-river hydropower scheme commenced abstraction in October 2008 and has a capacity of 0.25 MW. The Kinnaird Burn flows through nutrient poor moor land in its upper reaches before descending through a steep valley to the confluence with the River Tummel, a tributary of the River Tay. The Kinnaird Burn hydropower scheme has two intakes at NN 95895985 and NN 95815987 and an outfall at NN 95165840 approximately 1 km upstream of the confluence with the River Tummel (Table 3.2; Figures 3.2 & 3.3).

Fisheries surveys were carried out annually at four sites in Kinnaird Burn between 2005 and 2011; three additional sites were surveyed annually between 2008 and 2011. The hydropower scheme commenced operation in October 2008 shortly after the fisheries surveys in September 2008. Due to flood damage to a fish pass in the lower reaches of the burn, the area of the proposed hydropower scheme is currently considered inaccessible to migratory salmonids.

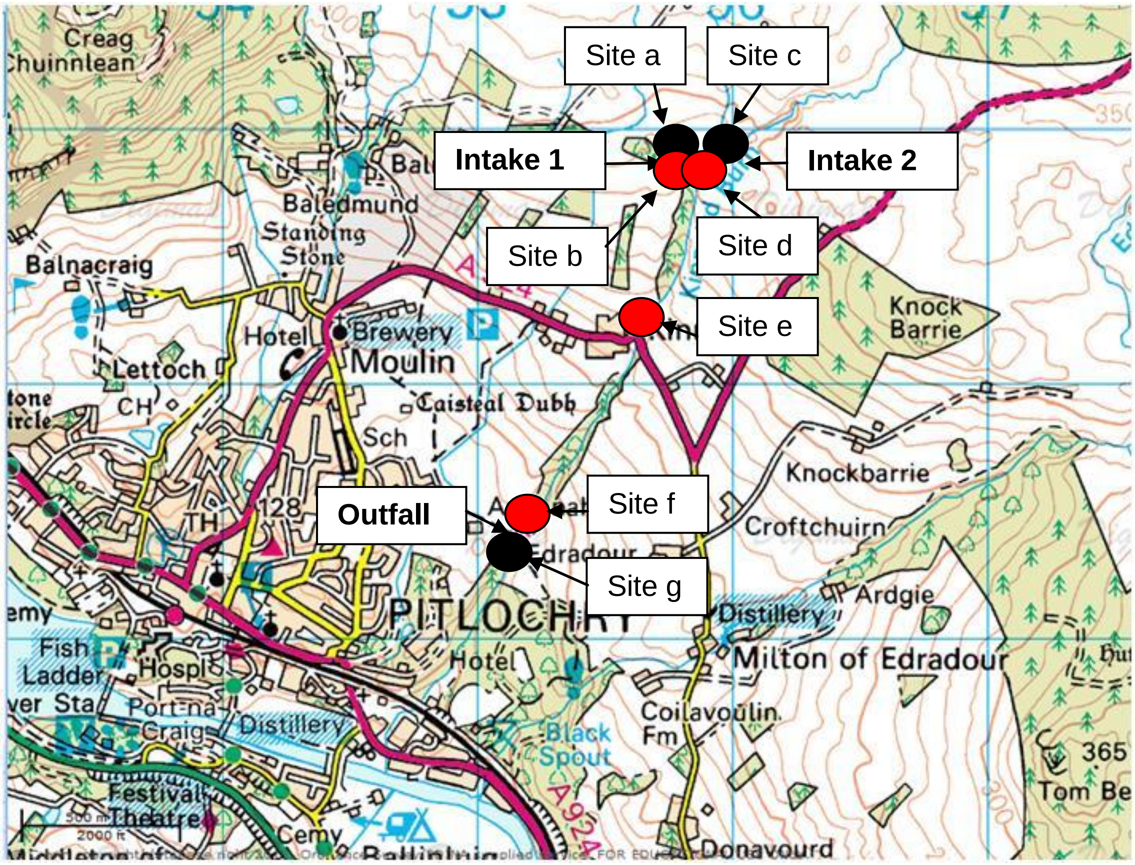

Figure 3.2 Location of intakes, outfall and survey sites on the Kinnaird Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of each intake and one downstream of the outfall).

(a) (b) (c)

Figure 3.3 Intake 1 (photo a), 2 (photo b) and outfall (photo c) structure at Kinnaird Burn.

Modelled flow data indicated the scheme would divert an increasingly greater proportion of the water above Q_{71} which increases to a maximum at about Q_{20}. The proportion of flow diverted progressively reduces at higher flows and as the flow regime approaches Q_{71} at which point abstraction ceases. Off-take weirs were built at each intake and the intake and outfall structures were guarded by 10 mm mesh screens to prevent entrainment of juvenile salmonids (Figure 3.3). Detailed rates of abstraction for both intakes are documented in Table 3.3.

Table 3.2 Fisheries survey site details in Kinnaird Burn in September 2005-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN 95955991	Immediately upstream of intake pool on main river	13/09/11 22/09/10 07/09/09 21/09/08	30 m/2.8 m/83 m^2 30 m/2.9 m/87 m^2 50 m/2.0 m/100 m^2 50 m/2.6 m/130 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
b* NN 95895985	Downstream of intake, immediately upstream of west tributary	13/09/11 22/09/10 07/09/09 21/09/08 19/09/07 15/09/06 24/09/05	55 m/3.1 m/171 m^2 48 m/2.5 m/122 m^2 60 m/2.0 m/120 m^2 50 m/2.5 m/125 m^2 60 m/2.7 m/162 m^2 59 m/3.1 m/183 m^2 49 m/4.3 m/211 m^2	BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI
c NN 95815987	Immediately upstream of intake on west tributary.	10/09/11 22/09/10 07/09/09 21/09/08	34 m/1.4 m/46 m^2 22 m/1.1 m/25 m^2 25 m/1.4 m/35 m^2 30 m/2.9 m/87 m^2	BP / Q / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
d* NN 95815987	Immediately downstream of intake on west tributary.	10/09/11 22/09/10 07/09/09 21/09/08	30 m/2.0 m/60 m^2 25 m/0.9 m/23 m^2 40 m/0.8 m/32 m^2 30 m/0.4 m/12 m^2	BP / Q / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
e* NN 95645923	Downstream of intake, above A924 road bridge	13/09/11 22/09/10 14/09/09 21/09/08 19/09/07 15/09/06 24/09/05	48 m/4.0 m/192 m^2 46 m/3.7 m/168 m^2 48 m/3.2 m/154 m^2 50 m/4.0 m/200 m^2 45 m/4.0 m/180 m^2 46 m/4.4 m/202 m^2 40 m/3.3 m/132 m^2	BP / Q / HIFI BP / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI
f* NN 95165840	Immediately upstream of outfall, upstream of footbridge at Auchnahyle	13/09/11 22/09/10 14/09/09 21/09/08 19/09/07 15/09/06 24/09/05	45 m/3.8 m/171 m^2 45 m/3.5 m/157 m^2 50 m/3.5 m/175 m^2 50 m/3.7 m/185 m^2 50 m/3.5 m/175 m^2 45 m/4.4 m/198 m^2 45 m/2.8 m/126 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI
g NN 95155834	Immediately downstream of outfall, downstream of footbridge at Auchnahyle	13/09/11 22/09/10 14/09/09 21/09/08 19/09/07 15/09/06 24/09/05	44 m/4.6 m/202 m^2 45 m/4.2 m/189 m^2 48 m/3.8 m/182 m^2 50 m/4.2 m/210 m^2 45 m/3.3 m/148 m^2 43 m/4.9 m/211 m^2 45 m/3.6 m/162 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI

Table 3.3 Rates of abstraction at Kinnaird Burn intakes.

NGR	Intake reference number	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off flow (L/s) (Q_{95})	Compensation flow (L/s) at maximum abstraction rate (Q_{82})
NN 95815987	Intake 1	63	5443.2	10	18
NN 95895985	Intake 2	147	12700	25	42

3.2.2 Keltney Burn

The Keltney Burn run-of-river hydropower scheme commenced abstraction in August 2010 and has a capacity of 2 MW; satisfying the needs of some 2000 average homes in Perthshire, encompassing those located within Keltney Burn village. The upper reaches of the Keltney Burn flow through nutrient poor moor land and heath land, before descending through a steep narrow gorge to the village of Keltney Burn. The narrow gorge and falls of Keltney act as an impassable barrier to migratory salmonids, thus they are restricted to the lower 500 m of the river where the gradient is shallow. Fisheries surveys were carried out annually at three sites in the lower reaches of Keltney Burn between 2005 and 2011; two sites in the upper reaches were surveyed annually between 2005 and 2007 and in 2010 and 2011, while one site in the upper reaches was surveyed in 2010 and 2011 (Table 3.4; Figure 3.4).

Table 3.4 Fisheries survey site details in Keltney Burn in September 2005-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN 75125333	Immediately upstream of intake	15/09/11	54 m/9.3 m/500 m^2	BP / SQ / HIFI
		13/09/10	65 m/9.7 m/631 m^2	BP / SQ / HIFI
b* NN 75205320	Downstream of intake upstream of site c	15/09/11	42 m/8.7 m/368 m^2	BP / SQ / HIFI
		13/09/10	40 m/8.7 m/348 m^2	BP / SQ / HIFI
		09/09/07	50 m/7.1 m/355 m^2	BP / SQ / HIFI
		14/09/06	42 m/5.0 m/210 m^2	BP / SQ / HIFI
		22/09/05	50 m/7.1m/355 m^2	BP / SQ / HIFI
c* NN 75505296	Above deer park, downstream of intake	15/09/11	50 m/7.3 m/365 m^2	BP / SQ / HIFI
		13/09/10	40 m/7.1 m/284 m^2	BP / SQ / HIFI
		09/09/07	64 m/6.0 m/384 m^2	BP / SQ / HIFI
		14/09/06	50 m/8.1 m/405 m^2	BP / SQ / HIFI
		22/09/05	50 m/8.1 m/405 m^2	BP / SQ / HIFI
d* NN 77434905	Cottage, upstream of outfall	15/09/11	40 m/9.6 m/384 m^2	G / Q / HIFI
		07/09/10	40 m/9.3 m/372 m^2	G / Q / HIFI
		05/09/09	45 m/10.5 m/472 m^2	G / Q / HIFI
		07/09/08	54 m/8.4 m/453 m^2	G / Q / HIFI
		04/09/07	50 m/8.1 m/405 m^2	G / Q / HIFI
		18/09/06	50 m/9.1 m/455 m^2	G / Q / HIFI
		21/09/05	50 m/8.2 m/410 m^2	G / Q / HIFI
e* NN 77424896	Upstream 2nd glide above River Lyon confluence to torrent. Upstream of outfall	15/09/11	55 m/8.8 m/484 m^2	G / Q / HIFI
		07/09/10	48 m/8.6 m/413 m^2	G / Q / HIFI
		14/09/09	50 m/8.3 m/415 m^2	G / Q / HIFI
		07/09/08	50 m/7.7 m/385 m^2	G / Q / HIFI
		04/09/07	53 m/6.9 m/366 m^2	G / Q / HIFI
		05/09/06	50 m/7.7 m/385 m^2	G / Q / HIFI
		21/09/05	50 m/6.9 m/345 m^2	G / Q / HIFI
f NN 77374885	River Lyon confluence to riffle at top of junction pool (at outfall)	15/09/11	40 m/12.9 m/516 m^2	G / Q / HIFI
		07/09/10	40 m/12.1 m/484 m^2	G / Q / HIFI
		05/09/09	40 m/12.1 m/484 m^2	G / Q / HIFI
		07/09/08	40 m/11.1 m/444 m^2	G / Q / HIFI
		04/09/07	40 m/10.9 m/436 m^2	G / Q / HIFI
		05/09/06	40 m/11.0 m/440 m^2	G / Q / HIFI
		21/09/05	33 m/12.1 m/399 m^2	G / Q / HIFI

The hydropower scheme commenced operation in August 2010 shortly before the fisheries surveys in September 2010. The Keltney Burn hydropower scheme has an intake at NN 75125333, and outfall at NN 77374887 close to the confluence of Keltney Burn with the River Lyon (Figures 3.4 & 3.5).

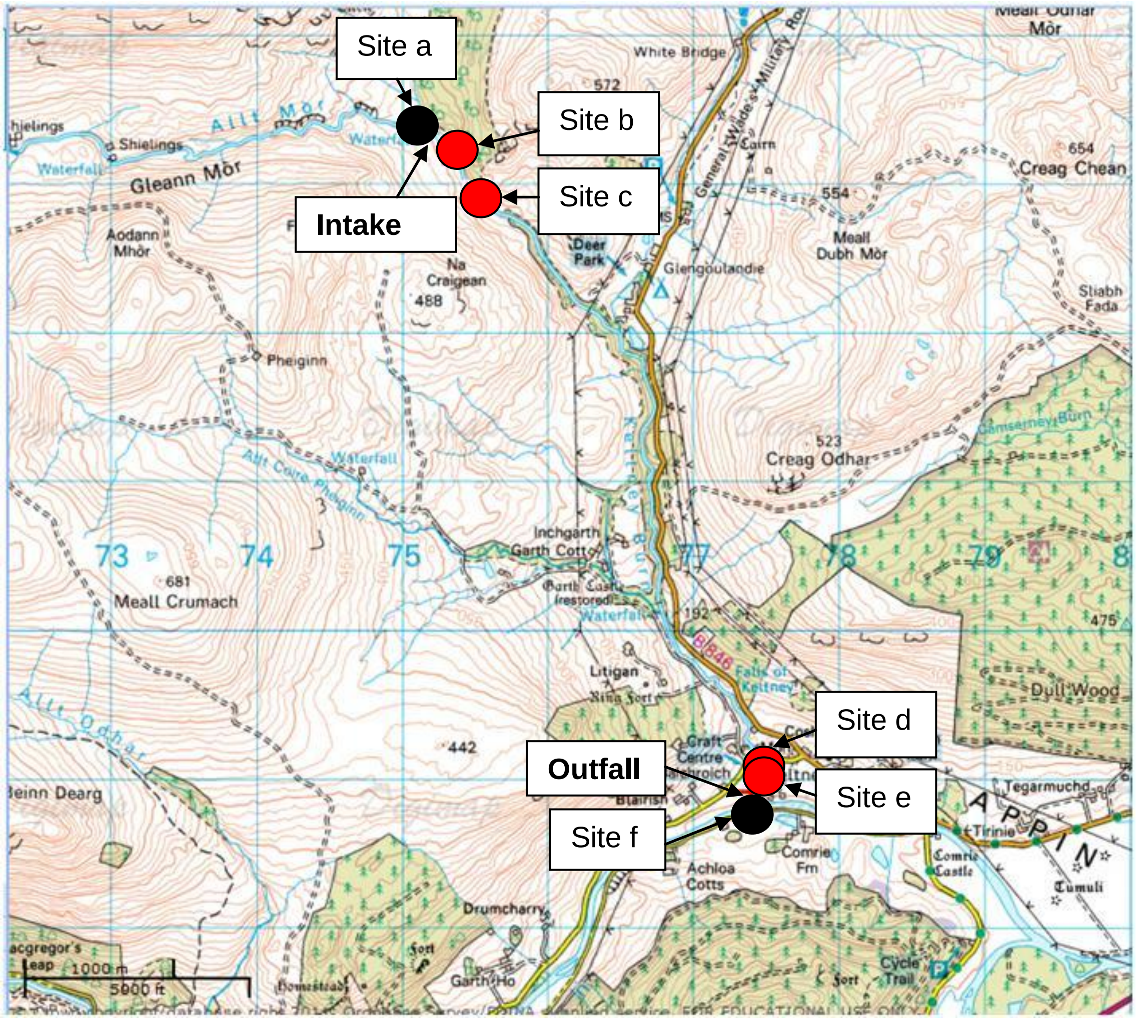

Figure 3.4 Location of intake, outfall and survey sites on the Keltney Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of the intake and one downstream of the outfall).

An off-take weir was built at the intake, and the intake and outfall structures were guarded by 10 mm mesh screens to prevent entrainment of juvenile salmonids. Modelled flow data indicated the scheme would divert an increasingly greater proportion of the water above Q_{85} to a maximum at about Q_{30}. The proportion of flow diverted progressively reduces at higher flows and as the flow regime approaches Q_{85} at which point abstraction ceases. Rates of abstraction are documented in Table 3.5.

(a)

(b)

Figure 3.5 Intake (photo a) and outfall (photo b) structure at Keltney Burn.

Table 3.5 Rates of abstraction at Keltney Burn.

NGR	Intake reference	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off Flow (L/s)	Residual flow (L/s) at maximum abstraction rate
NN 75125333	Intake	1000	86 400	160	300

3.2.3 Rottal Burn

The run-of-river hydropower scheme in Rottal Burn commenced abstraction in December 2008 and has a generating capacity capable of providing electricity to an estimated 500 homes. Fisheries surveys were carried out at six sites in 2006 and 2007 and annually between 2009 and 2011; one site was surveyed in 2007 and annually between 2009 and 2011 (Table 3.6; Figure 3.6). The hydropower scheme commenced abstraction in December 2008.

Table 3.6 Fisheries survey site details in Rottal Burn in September 2006-2007 and 2009-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NO 38187224	Burn of Heughs, 50 m upstream of intake	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	40 m/4.2 m/168 m^2 40 m/3.3 m/132 m^2 46 m/3.7 m/170 m^2 45 m/4.0 m/180 m^2 46 m/4.5 m/207 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
b* NO 37316992	Burn of Heughs, downstream of intake; adjacent to lunch hut	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	40 m/4.0 m/160 m^2 40 m/3.5 m/140 m^2 46 m/4.0 m/184 m^2 45 m/3.8 m/171 m^2 43 m/3.9 m/167 m^2	BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI
c NO 39087142	Kennel Burn, 20 m upstream of intake	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	40 m/3.4 m/137 m^2 40 m/2.4 m/96 m^2 45 m/3.7 m/167 m^2 55 m/2.5 m/138 m^2 40 m/3.0 m/120 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
d* NO 38867125	Kennel Burn, 60m downstream of intake	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	38 m/2.0 m/76 m^2 40 m/2.2 m/88 m^2 45 m/2.6 m/117 m^2 50 m/3.2 m/160 m^2 50 m/2.7 m/135 m^2	BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / SQ / HIFI BP / SQ / HIFI
e* NO 37617036	Burn of Heughs, immediately downstream of old bridge crossing river	05/09/11 06/09/10 13/09/09 07/09/07	40 m/3.1 m/124 m^2 40 m/4.4 m/176 m^2 40 m/5.4 m/216 m^2 40 m/6.1 m/244 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
f* NO 37677148	Rottal Burn, 10 m upstream of outfall	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	40 m/6.3 m/252 m^2 40 m/5.3 m/212 m^2 42 m/6.3 m/265 m^2 44 m/7.0 m/308 m^2 40 m/6.7 m/268 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI
g NO 37266986	Rottal Burn, upstream road bridge at lodge, 70 m downstream of outfall	05/09/11 06/09/10 13/09/09 07/09/07 08/09/06	35 m/6.1 m/214 m^2 40 m/5.9 m/236 m^2 40 m/6.0 m/240 m^2 40 m/5.2 m/208 m^2 46 m/5.6 m/258 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI

The Rottal Burn, a tributary of the River South Esk, is formed by two tributaries, the Burn of Heughs and the Kennel Burn; as such the scheme has two intakes at NO 38157215 (Burn of Heughs) and NO 39127144 (Kennel Burn) both encompassing a small off-take weir. The outfall at NO 37276986 is located upstream of the confluence with the River South Esk (Table 3.6; Figure 3.6) and was designed to prevent local erosion of the river bed; furthermore the discharge and positioning of the outfall ensured adult migratory fish would not be distracted from the main flow moving into Rottal Burn due to the low efflux velocity. Both the intakes and outfalls are guarded by 10 mm mesh screens to prevent entrainment of fish (Figure 3.7).

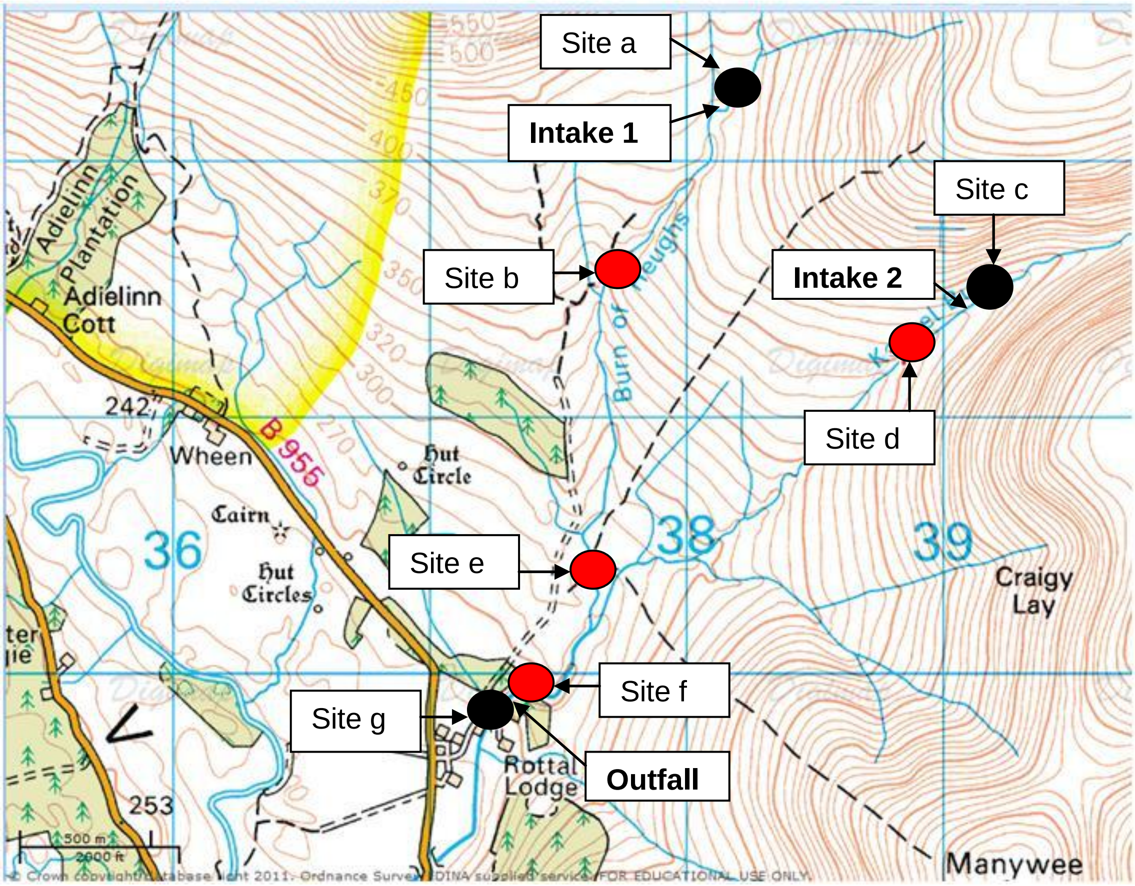

Figure 3.6 Location of intakes, outfall and survey sites on Rottal Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of each intake and one downstream of the outfall).

(a) (b) (c)

Figure 3.7 Intakes 1 (photo a), 2 (photo b) and outfall (photo c) structure at Rottal Burn.

The scheme is designed to divert an increasingly greater proportion of the water above Q_{85} to a maximum at about Q_{20}. Detailed rates of abstraction are documented in Table 3.7. It is important to note that the Rottal Burn hydropower scheme was out of operation for some periods of 2011; the reasons for this are unknown.

Table 3.7 Rates of abstraction at Rottal Burn.

NGR	Intake reference number	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off Flow (L/s) (Q_{88})	Compensation flow (L/s) at maximum abstraction rate (Q_{51})
NO 38157215	Intake 1	212	18316	72	120
NO 39127144	Intake 2	105	9072	34	60

3.2.4 Innerhadden Burn

The Innerhadden Burn run-of-river hydropower scheme commenced abstraction in November 2009 and has a generating capacity of 1.4 MW; whilst the scheme is generally small it is capable of supporting around 1000 homes in Perthshire. The Innerhadden Burn is formed by two tributaries, Glen Sassunn Burn and Allt Coire Cruach Sneachda and thus has two intakes at NN 65505408 (Glen Sassunn Burn) and NN 67405360 (Allt Coire Cruach Sneachda); a 6 km pipeline leads from its two intakes down to the turbine house near the outfall at NN 67365757 (Table 3.8; Figure 3.8).

The river flows through nutrient poor moor land before descending through a steep gorge in deciduous woodland to Innerhadden. The burn then flows through rough pastureland in the Kinloch Rannoch valley for approximately 1 km before joining Dunalastair Water; the latter water body flows into Loch Tummel and subsequently the River Tummel with the confluence with the River Tay at Ballinluig. Migratory salmonids can access Dunalastair Water, but they first must traverse the fish passes at Pitlochry and Clunie Dam, and access in Innerhadden Burn is restricted to the lower reaches downstream of the gorge and waterfall.

Fisheries surveys were carried out at five sites in Innerhadden Burn in September 2005 and 2006, six sites in September 2007, two sites in the lower reaches in September 2008 and six sites in September 2010 and 2011. The hydropower scheme commenced abstraction in November 2009.

Modelled flow data indicated the scheme would divert an increasingly greater proportion of the water above Q_{85} which increases to a maximum at about Q_{20}. The proportion of flow diverted progressively reduces at higher flows and as the flow regime approaches Q_{85} at which point abstraction ceases. Detailed rates of abstraction are documented in Table 3.9. Off-take weirs were built at each intake and the intake and outfall structures were guarded by 10 mm mesh screens to prevent entrainment of juvenile salmonids (Figure 3.9).

Table 3.8 Fisheries survey site details in Innerhadden Burn in September 2005-2008 and 2010-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN 65385405	Glen Sassunn Burn – upstream of intake	04/09/11 08/09/10 08/09/07 13/09/06 18/09/05	40 m/3.4 m/136 m^2 45 m/3.6 m/162 m^2 60 m/3.3 m/198 m^2 60 m/4.5 m/270 m^2 49 m/2.9 m/142 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
b* NN 65505408	Glen Sassunn Burn – downstream of intake	04/09/11 08/09/10 08/09/07 13/09/06 18/09/05	42 m/6.1 m/256 m^2 40 m/4.0 m/160 m^2 40 m/6.2 m/248 m^2 50 m/6.1 m/305 m^2 41 m/6.3 m/258 m^2	BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / Q / HIFI BP / SQ / HIFI
c NN 67355381	Allt Coire Cruach Sneachda – upstream of intake	04/09/11 08/09/10 08/09/07 13/09/06 18/09/05	50 m/4.3 m/215 m^2 45 m/4.4 m/198 m^2 70 m/2.3 m/161 m^2 50 m/3.5 m/175 m^2 50 m/3.8 m/190 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
d* NN 67265407	Allt Coire Cruach Sneachda – downstream of intake	04/09/11 08/09/10 08/09/07 13/09/06 18/09/05	40 m/3.8 m/152 m^2 45 m/4.6 m/207 m^2 50 m/3.2 m/160 m^2 50 m/4.6 m/230 m^2 50 m/6.0 m/300 m^2	BP / Q / HIFI BP / Q / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / Q / HIFI
e* NN 67275758	Adjacent to island, upstream of outfall	04/09/11 08/09/10 15/09/08 09/09/07 14/09/06 22/09/05	45 m/9.4 m/423 m^2 43 m/10.1 m/434 m^2 40 m/10.2 m/408 m^2 50 m/9.4 m/470 m^2 40 m/9.7 m/388 m^2 50 m/10.8 m/540 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI
f NN 67365757	50m downstream of outfall	04/09/11 08/09/10 15/09/08 09/09/07	40 m/8.5 m/340 m^2 40 m/8.6 m/344 m^2 40 m/9.5 m/380 m^2 60 m/9.1 m/546 m^2	G / Q / HIFI G / Q / HIFI G / Q / HIFI G / Q / HIFI

Table 3.9 Rates of abstraction at Innerhadden Burn.

NGR	Intake reference number	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off Flow (L/s)	Compensation flow (L/s) at maximum abstraction rate (Q_{82})
NN 65505408	Glen Sassunn (Intake 1)	350	30240	70	200
NN 67405360	Allt Coire Cruach Sneachda (Intake 2)	200	17280	40	100

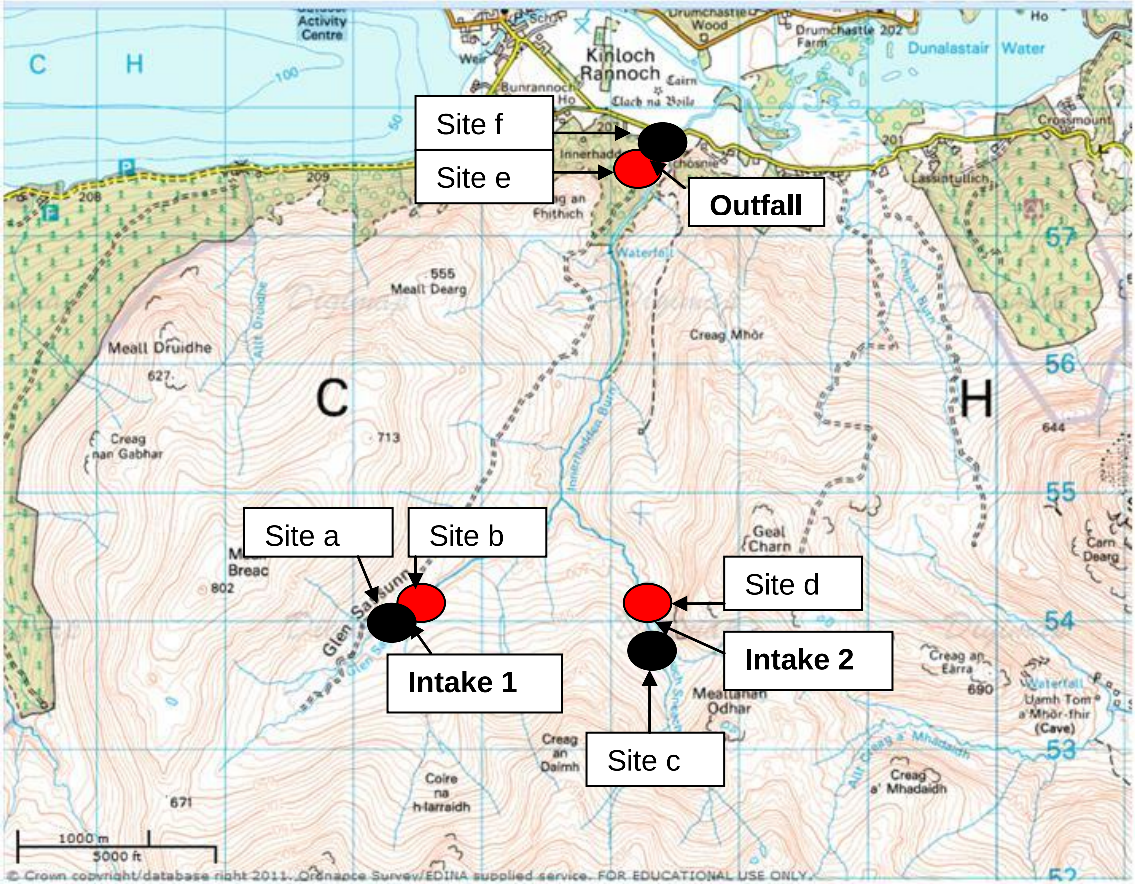

Figure 3.8 Location of intakes, outfall and survey sites on Innerhadden Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of each intake and one downstream of the outfall).

(a)

(b)

(c)

Figure 3.9 Intakes 1 (photo a), 2 (photo b) and outfall (photo c) structure at Innerhadden Burn.

3.2.5 Inverhaggernie Burn

The Inverhaggernie Burn run-of-river hydropower scheme commenced abstraction in October 2009 and has a generating capacity of 680 kW, the scheme was one of the first hydro planning applications within the Loch Lomond & Trossachs National Park. Inverhaggernie Burn, located in Crianlarich, Perthshire, is a tributary of the River Fillan, the latter becoming the River Dochart before flowing into Loch Tay. The Inverhaggernie Burn hydropower scheme has an intake at NN 38652855 and outfall at NN 37332684 approximately 150 m upstream of the confluence with the River Fillan (Table 3.10; Figure 3.10). Fisheries surveys were carried out at three sites in Inverhaggernie Burn in September 2006, two sites in September 2007, and four sites in September 2008, 2010 and 2011. The hydropower scheme commenced abstraction in October 2009.

Modelled flow data indicated the scheme would divert an increasingly greater proportion of the water above Q_{85} to a maximum at about Q_{20}. The proportion of flow diverted progressively reduces at higher flows and as the flow regime approaches Q_{85} at which point abstraction ceases. Rates of abstraction are detailed in Table 3.11. It is important to note that the Inverhaggernie Burn hydropower scheme was out of operation for some periods of autumn and early winter of 2011; the reasons for this are unknown. An off-take weir was built at the intake and the intake and outfall structures were guarded by 10 mm mesh screens to prevent entrainment of juvenile salmonids (Figure 3.11).

Table 3.10 Fisheries survey site details in Inverhaggernie Burn in September 2006-2008 and 2010-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN 38542843	First pool upstream of intake	16/09/11	45 m/4.5 m/203 m^2	BP / Q / HIFI
		10/09/10	45 m/4.8 m/216 m^2	BP / SQ / HIFI
		06/09/08	55 m/5.8 m/319 m^2	BP / SQ / HIFI
		12/09/06	60 m/4.75 m/285 m^2	BP / SQ / HIFI
b* NN 38482833	200 m downstream of intake	16/09/11	40 m/4.9 m/196 m^2	BP / Q / HIFI
		10/09/10	45 m/5.0 m/225 m^2	BP / Q / HIFI
		06/09/08	50 m/4.4 m/220 m^2	BP / Q / HIFI
		12/09/06	56 m/5.4 m/303 m^2	BP / Q / HIFI
c* NN37402691	10 m upstream of bridge, upstream of outfall	16/09/11	40 m/6.6 m/264 m^2	G / Q / HIFI
		10/09/10	40 m/7.4 m/296 m^2	G / Q / HIFI
		06/09/08	44 m/5.4 m/238 m^2	G / Q / HIFI
		12/09/07	47 m/6.5 m/306 m^2	G / Q / HIFI
		12/09/06	50 m/6.6 m/330 m^2	G / Q / HIFI
d NN37332684	10 m downstream of outfall	16/09/11	40 m/6.8 m/272 m^2	G / Q / HIFI
		10/09/10	41 m/7.0 m/287 m^2	G / Q / HIFI
		06/09/08	45 m/5.1 m/243 m^2	G / Q / HIFI
		12/09/07	40 m/5.3 m/330 m^2	G / Q / HIFI

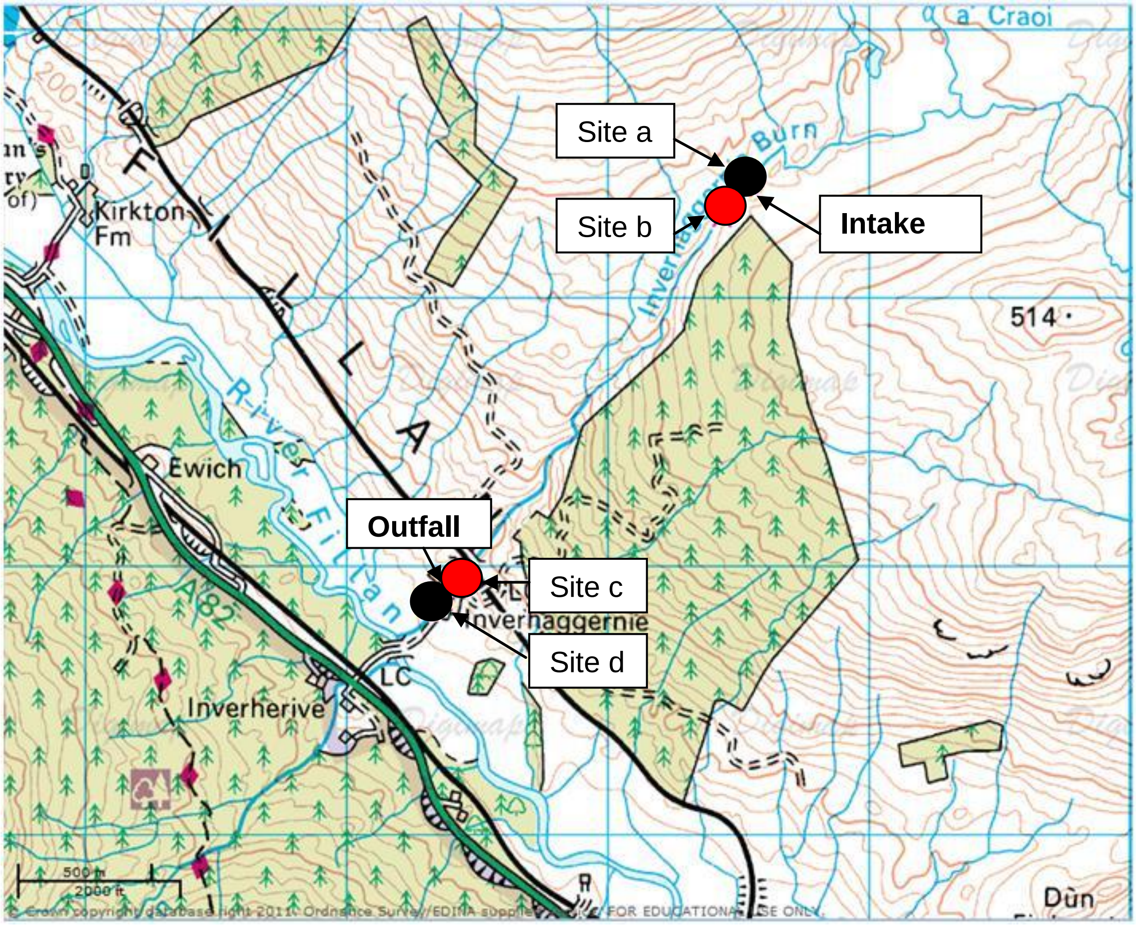

Figure 3.10 Location of intake, outfall and survey sites on Inverhaggernie Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of the intake and one downstream of the outfall).

(a) (b)

Figure 3.11 Intake (photo a) and outfall (photo b) structure at Inverhaggernie Burn.

Table 3.11 Rates of abstraction at Inverhaggernie Burn.

NGR	Intake reference	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off Flow (L/s)	Residual flow (L/s) at maximum abstraction rate
NN38652855	Intake	383	33091	46	70

3.2.6 River Callop

The River Callop flows through nutrient poor moor land and forestry before descending to a shallow valley, eventually flowing in to Loch Shiel. The hydropower scheme commenced abstraction in 2008 and has a generating capacity of 1087 kW. The River Callop is formed from four tributaries and as such has four intakes at NM 91447764 (Allt an Fhaing), NM 91857701 (Allt na-Cruaiche), NM 92477652 (Allt Coire Fada) and NM 92557652 (Allt á Choire Chruinn); the outfall is located at NM 92507878 (Table 3.12; Figure 3.12). Off-take weirs were built at each intake and intake and outfall structures were guarded by mesh screens to prevent entrainment of juvenile salmonids (Figure 3.13). Intakes and survey sites use the allocated numbers/site names as given by MFC and thus were adopted in their original format for inclusion in this study.

Table 3.12 Fisheries survey site details in the River Callop in September 2006 and 2010-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute, MFC – Morgan Fisheries Consultancy, LFT – Lochaber Fisheries Trust) $^{+}$ mean river width not known, $^{++}$ NGR supplied did not match up to a river, $^{+++}$ survey date not known, $^{++++}$ area surveyed not known. Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
c5* NM92067734	A short distance downstream from intake 3, in lower gradient area at top of small woodland on left bank	08/09/11 17/09/10 26/09/06	30 m/6.0 m/180 m^2 30 m/5.2 m/156 m^2 53 m/543 m^{2+}	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / MFC
c3* NM91717786	Downstream intake 4. Alongside 3 birches on RB, start at top of island	08/09/11 17/09/10 26/09/06	35 m/3.1 m/108 m^2 32 m/2.9 m/93 m^2 34 m/82 m^{2+}	BP / Q / HIFI BP / Q / HIFI BP / SQ / MFC
c6* NM92457803	Bottom of site is 75 m upstream of confluence	08/09/11 17/09/10 27/09/06	38 m/4.6 m/175 m^2 30 m/3.3 m/99 m^2 20 m/107 m^{2+}	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / MFC
c7* NM92427810	20 m upstream from confluence	08/09/11 17/09/10 27/09/06	27 m/5.9 m/159 m^2 30 m/5.6 m/168 m^2 18 m/146 m^{2+}	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / MFC
c2* NM92427829	Top of site is small weir 8 m upstream tributary	08/09/11 17/09/10 03/09/06	31 m/4.6 m/143 m^2 30 m/4.7 m/141 m^2 29 m/100 m^{2+}	BP / Q / HIFI BP / Q / HIFI BP / Q / MFC
c4*($^{++}$)	120 m upstream of Allt an Fhaing confluence	26/09/06 2010^{+++}	26 m/166 m^{2+} $^{++++}$	BP / Q / MFC BP / SQ / LFT
cH NM92507878	10 m downstream of outfall, upstream of site c1	08/09/11 17/09/10	37 m/9.9 m/366 m^2 37 m/7.8 m/289 m^2	BP / Q / HIFI BP / Q / HIFI
c1 NM92497908	Downstream proposed powerhouse. Top of site 12 m downstream of alders	08/09/11 2010^{+++} 13/09/06	30 m/7.1 m/213 m^2 $^{++++}$ 28 m/179 m^{2+}	BP / Q / HIFI BP / SQ / LFT BP / Q / MFC

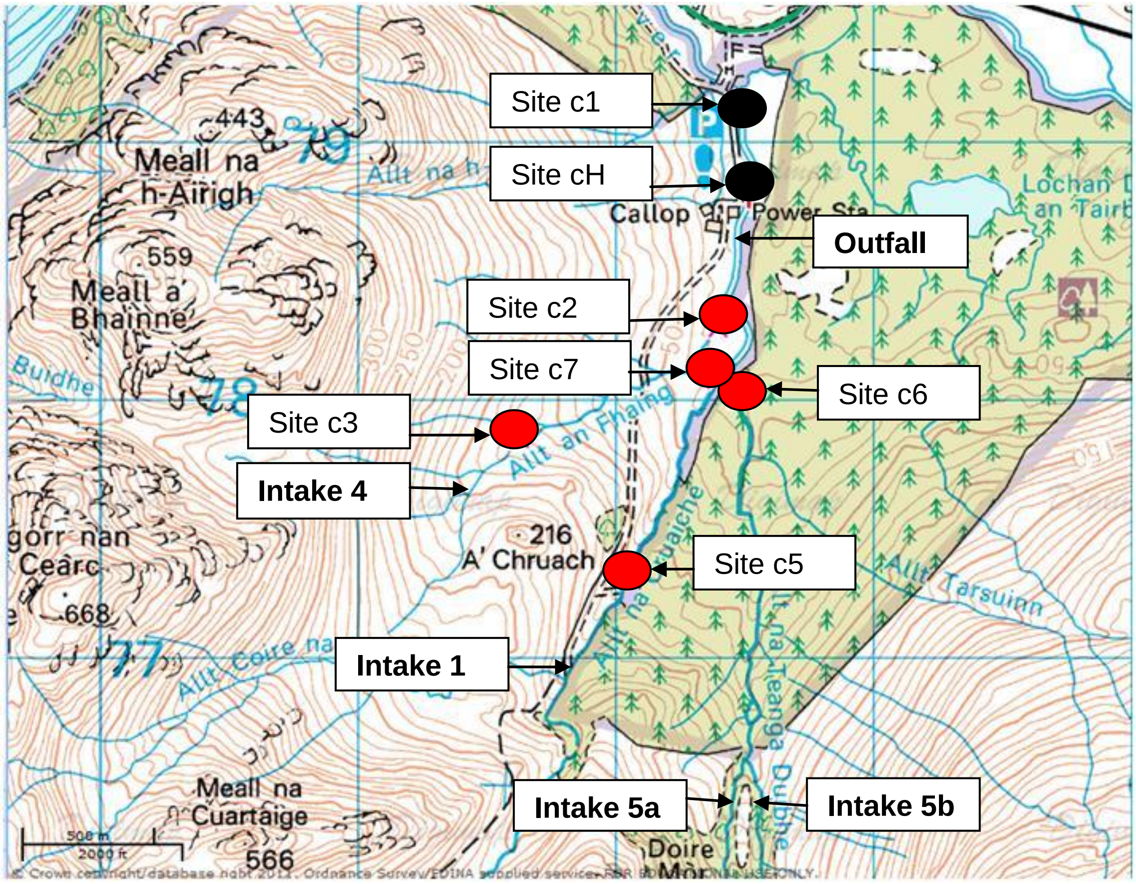

Figure 3.12 Location of intakes, outfall and survey sites on the River Callop (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in the lower reaches situated outside of the depleted stretch, downstream of the outfall).

(a) (b)

Figure 3.13 Intake (a) and outfall (b) structure on the River Callop. (Photo courtesy of Kjersti Birkeland, Anthony Watkins and John Webb)

Fisheries surveys were carried out in 2006 by Morgan Fisheries Consultancy (MFC), prior to the commencement of operation of the scheme in 2008. Fisheries surveys were carried out by the author in 2010 and 2011 after the scheme was operational; additionally the Lochaber Fisheries Trust (LFT) carried out surveys in 2010 (Table 3.17). No modelled flow data were available for the scheme for this report but rates of abstraction for each of intake are documented in Table 3.13.

Table 3.13 Rates of abstraction at the River Callop.

NGR	Intake reference number	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off flow (L/s)
NM 91857701	Intake 1	672	58300	36
NM 91447764	Intake 4	132	11400	5
NM 92477652	Intake 5a	198	17150	6
NM 92557652	Intake 5b	198	17150	10

3.2.7 Ardvorlich Burn

The Ardvorlich Burn run-of-river hydropower scheme, located along the south shores of Loch Earn, Perthshire has a generating capacity of 716 kW. The scheme comprises three intakes, one situated on each of the three tributaries that form the Ardvorlich Burn; a 2.7 km pipe transfers water from each of the intakes to the turbine house. The Ardvorlich Burn flows through nutrient poor moor land in its upper reaches before descending through a steep gorge to the valley floor and on to the confluence with Loch Earn. The Ardvorlich Burn hydropower scheme has three intakes at NN 630211, NN 635209 and NN 636209 and an outfall at NN 632228 (Table 3.14; Figure 3.14).

Fisheries surveys were carried out in 2008 by MFC, prior to the commencement of water abstraction by the scheme in 2011; surveys were carried out in 2011 by SEPA with the scheme operational (Table 3.14). No ageing or HABSCORE data were collected in the surveys by MFC or SEPA. Intakes and survey sites use the allocated numbers/site names as given by MFC and SEPA and thus were adopted in their original format for inclusion in this study.

Off-take weirs were built at each intake, and intake and outfall structures were guarded by mesh screens to prevent entrainment of juvenile salmonids (note no photographs of intake/outfall locations available). No modelled flow data were available for the scheme for this report but detailed rates of abstraction are documented in Table 3.15.

Table 3.14 Fisheries survey site details in Ardvorlich Burn in August 2008 and September 2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, SEPA – Scottish Environment Protection Agency, MFC – Morgan Fisheries Consultancy) $^{+}$ mean river width not known. Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
A6 NN 6299 2116	Upstream of track (from 40-70 m upstream of bridge), Upstream of intake 5	07/09/11 02/07/08	20.5 m/5.2 m/107 m^2 30 m/110 m^{2+}	BP / SQ / SEPA BP / SQ / MFC
A5* NN 6332 2134	Upstream of Allt a' Choire Bhuide (bottom of site was 5 m above the deplanked bridge). Downstream of intakes 6a/6b	07/09/11 02/07/08	18 m/5.6 m/101 m^2 15 m/58 m^{2+}	BP / SQ / SEPA BP / SQ / MFC
A4* NN 6313 2161	Top of site approx 10 m downstream of small tributary on left bank. Downstream of all intakes	07/09/11 02/07/08	21.5 m/5.5 m/118 m^2 12 m/91 m^{2+}	BP / SQ / SEPA BP / SQ / MFC
A3* NN 6322 2290	Downstream of footbridge (for 18 m upstream to small fall under bridge). Upstream of outfall	08/09/11 02/07/08	12.5 m/5.4 m/68 m^2 18 m/103 m^{2+}	BP / SQ / SEPA G / SQ / MFC
A2 NN 6326 2302	In field downstream of roadbridge. Downstream of outfall	08/09/11 02/07/08	20 m/5.8 m/116 m^2 18 m/103 m^{2+}	BP / SQ / SEPA G / Q / MFC
A1 NN 6330 2318	In field downstream of roadbridge, downstream of site A2. Downstream of outfall	08/09/11 02/07/08	29 m/5.4 m/157 m^2 18 m/103 m^{2+}	BP / Q / SEPA G / Q / MFC

Table 3.15 Rates of abstraction at Ardvorlich Burn.

NGR	Intake reference number	Maximum abstraction rate (m^3/s)	Maximum daily volume (m^3/d)	Hands-off Flow (m^3/s)
NN 630211	Intake 5	0.2456	21220	0.015
NN 635209	Intake 6a	0.079	6826	0.011
NN 636209	Intake 6b	0.1	8640	0.006

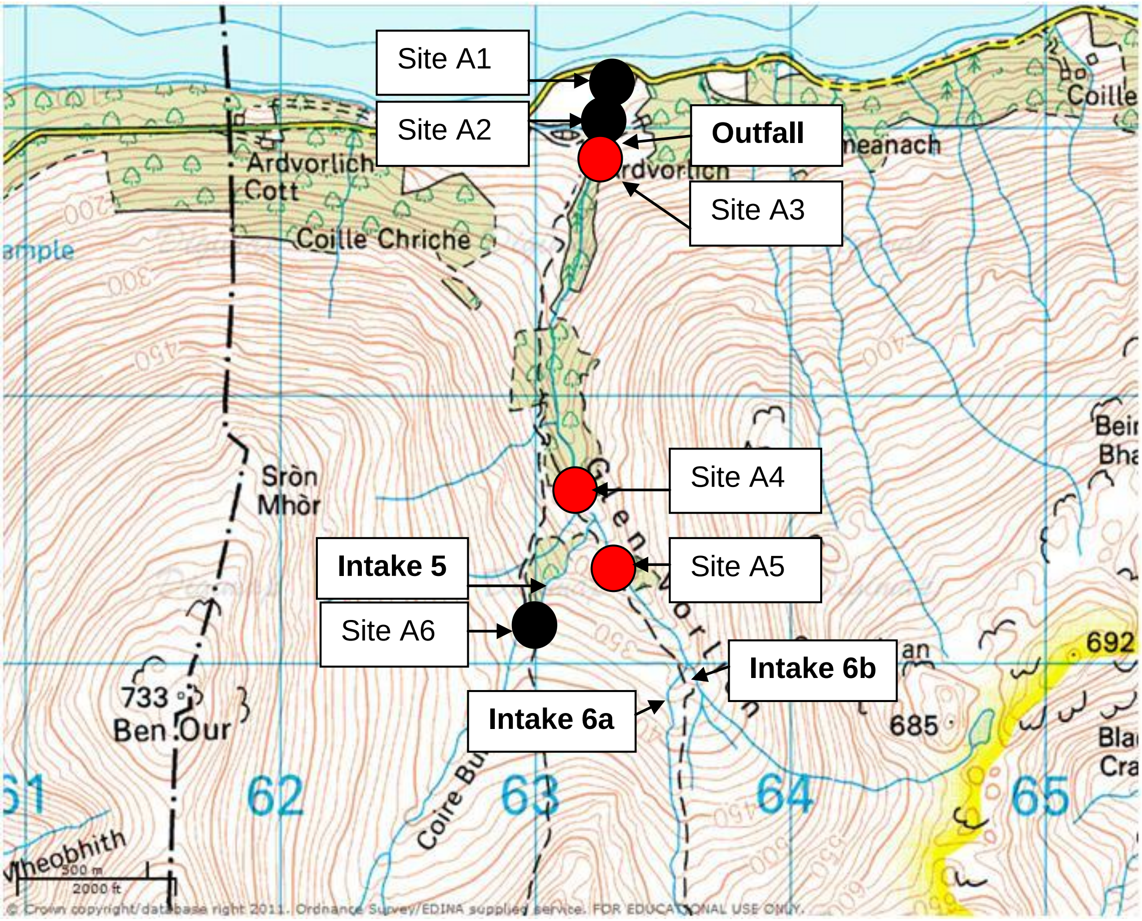

Figure 3.14 Location of intakes, outfall and survey sites on Ardvorlich Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of one intake and two downstream of the outfall).

3.2.8 Douglas Water

The Douglas Water run-of-river hydropower scheme is situated above the western shores of Loch Fyne, flowing down steep sloping hillside in a rural location; it has an estimated generating capacity of 3 MW. The scheme comprises two single intake weirs, one located on Douglas Water and the other on one of its tributaries, Allt Fearna. The land through which the river flows is comprised of a combination of commercial forestry, river corridor habitats and broadleaved woodland. There are no statutory landscape or nature conservation designations in the vicinity of the proposed development site and due to forestry paths and a former water storage reservoir located upstream of the power house, the land has been unnaturally modified and subsequently degraded. Douglas Water flows through forestry land throughout its length descending to the confluence with Loch Fyne. The Douglas Water hydropower scheme has two intakes at NN 04020758 and NN 04030697 and an outfall at NN 05370491 (Table 3.16; Figure 3.15). Fisheries surveys were carried out in 2002 by MFC, prior to the commencement of water abstraction by the scheme in April 2008; surveys were carried out in 2011 by SEPA with the scheme operational. Intakes and survey sites use the allocated numbers/site names as given by MFC and SEPA and thus were adopted in their original format for inclusion in this study.

Table 3.16 Fisheries survey site details in Douglas Water in 2002 and October 2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, SEPA – Scottish Environment Protection Agency, MFC – Morgan Fisheries Consultancy) + survey equipment unknown. Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
D8 NN 0384 0757	Upstream intake	04/10/11 2002	119.3 m/10.6 m/205 m^2 9.4 m/11.8 m/230 m^2	BP / SQ / SEPA +Q / MFC
D7* NN 0420 0710	Downstream Allt Nam Muc intake	03/10/11 2002	30.3 m/3.1 m/94 m^2 20.1 m/8.8 m/177 m^2	BP / SQ / SEPA +Q / MFC
D6* NN 0427 0729	Downstream of both intakes	04/10/11 2002	22.5 m/10.6 m/239 m^2 24.4 m/3.5 m/86 m^2	BP / Q / SEPA +Q / MFC
D5* NN 0542 0525	Downstream salmon ladder, upstream of outfall	03/10/11 2002	15.5 m/9.2 m/143 m^2 17.1 m/12.2 m/209 m^2	BP / SQ / SEPA +Q / MFC
D3 NN 0581 0477	Claonairigh Island, downstream of outfall	03/10/11 2002	18.5 m/9.1 m/168 m^2 15.3 m/8.9 m/136 m^2	BP / SQ / SEPA +Q / MFC

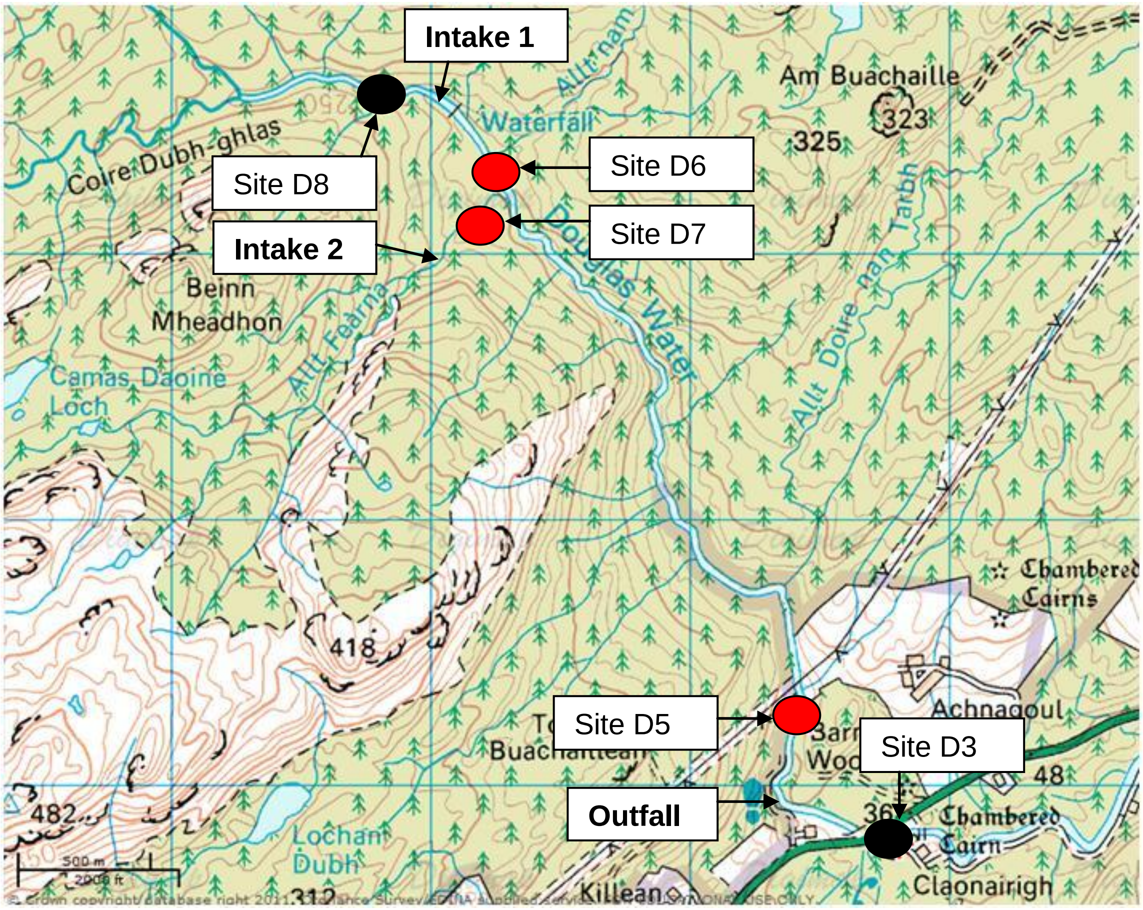

Figure 3.15 Location of intakes, outfall and survey sites on Douglas Water (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of one intake and one downstream of the outfall).

No modelled flow data were available for the scheme for this report. Off-take weirs were built at each intake and the intake and outfall structures were guarded by mesh screens to prevent entrainment of juvenile salmonids (note no photographs of intake/outfall locations available). It should be noted that the intake structure differs from many of the other schemes under assessment as the Douglas Water scheme utilises a coanda screen at the intake traversing the whole river width, whereas other schemes utilise screens set alongside the river channel. Detailed rates of abstraction are documented in Table 3.17. It should be noted that although Douglas Water run-of-river hydropower scheme has two intakes, rates of abstraction for only one of these were provided in the CAR licence.

Table 3.17 Rates of abstraction at Douglas Water.

NGR	Intake reference number	Maximum abstraction rate (L/s)	Maximum abstraction rate (m^3/d)	Hands-off Flow (L/s)
NN 3286 9773	Intake 1	2150	186 000	109

3.2.9 Camserney Burn

The Camserney Burn is a tributary of the River Tay, located in Aberfeldy, Perthshire. The run-of-river hydropower scheme was constructed in 2003 and subsequently became operational in 2004; the scheme has an estimated generating capacity of 900 kW. The upper reaches flow through coniferous woodland and moor land before descending through a steep gorge to Camserney village; the gorge acts as an impassable barrier to migratory salmonids. The last 1.5 km of the Camserney Burn flow through the River Tay floodplain and has a shallow gradient.

The Camserney Burn hydropower scheme has an intake at NN 80955045 and outfall at NN 81634934 upstream of the confluence with the River Tay (Table 3.18; Figure 3.16). Fisheries surveys were carried out annually between 2004 and 2011, with the scheme operational. Pre-operational fisheries data were not available at comparable sites in the reaches of the scheme. However the inclusion of control sites upstream of the intake and downstream of the outfall allow for identification of trends in populations under operation of the hydropower scheme.

The scheme was designed to divert a percentage of the water above Q_{85}, which increases to a maximum at about Q_{20} at which point abstraction ceases; rates of abstraction are documented in Table 3.19. An off-take weir was built at the intake and the intake and outfall structures were guarded by 10 mm mesh screens to prevent entrainment of juvenile salmonids (Figure 3.17) (note photograph of the outfall is unavailable). It should be noted that the Camserney Burn hydropower scheme had problems with low compensation flow in its early operation due to errors with the level of the compensation flow notch.

Table 3.18 Fisheries survey site details in Camserney Burn in September 2004-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN 80995044	Immediately upstream of intake	06/09/11	40 m/6.1 m/244 m^2	BP / SQ / HIFI
		05/09/10	40 m/4.8 m/194 m^2	BP / Q / HIFI
		07/09/09	45 m/5.1 m/229 m^2	BP / SQ / HIFI
		20/09/08	45 m/5.7 m/256 m^2	BP / Q / HIFI
		22/09/07	50 m/5.0 m/250 m^2	BP / SQ / HIFI
		17/09/06	50 m/4.9 m/245 m^2	BP / SQ / HIFI
		23/09/05	50 m/5.4 m/270 m^2	BP / SQ / HIFI
		04/09/04	50 m/5.8 m/290 m^2	BP / SQ / HIFI
b* NN 80935048	Immediately downstream of intake, upstream of gorge	06/09/11	50 m/4.0 m/200 m^2	BP / Q / HIFI
		05/09/10	50 m/4.5 m/225 m^2	BP / SQ / HIFI
		07/09/09	50 m/4.5 m/225 m^2	BP / Q / HIFI
		20/09/08	50 m/4.2 m/210 m^2	BP / SQ / HIFI
		22/09/07	50 m/4.5 m/225 m^2	BP / Q / HIFI
		17/09/06	50 m/4.4 m/220 m^2	BP / Q / HIFI
		23/09/05	60 m/4.5 m/270 m^2	BP / SQ / HIFI
		04/09/04	55 m/4.0 m/220 m^2	BP / SQ / HIFI
c* NN 81514955	Adjacent to public footpath upstream of outfall	07/09/11	53 m/4.9 m/260 m^2	G / Q / HIFI
		03/09/10	50 m/4.5 m/225 m^2	G / SQ / HIFI
		05/09/09	50 m/5.0 m/250 m^2	G / SQ / HIFI
		19/09/08	50 m/5.7 m/285 m^2	G / SQ / HIFI
		22/09/07	45 m/4.6 m/207 m^2	G / SQ / HIFI
		17/09/06	48 m/4.2 m/202 m^2	G / SQ / HIFI
		19/09/05	50 m/3.9 m/195 m^2	G / SQ / HIFI
		04/09/04	40 m/4.9 m/196 m^2	G / SQ / HIFI
d NN 81984918	Adjacent to village huts, downstream of outfall	07/09/11	40 m/5.3 m/212 m^2	G / Q / HIFI
		03/09/10	40 m/4.0 m/160 m^2	G / Q / HIFI
		05/09/09	40 m/5.4 m/216 m^2	G / Q / HIFI
		20/09/08	40 m/4.0 m/160 m^2	G / Q / HIFI
		20/09/07	45 m/4.0 m/180 m^2	G / Q / HIFI
		17/09/06	47 m/4.6 m/216 m^2	G / Q / HIFI
		19/09/05	50 m/3.7 m/185 m^2	G / Q / HIFI
		04/09/04	50 m/5.0 m/250 m^2	G / Q / HIFI

Table 3.19 Rates of abstraction at Camserney Burn.

NGR	Intake reference	Maximum abstraction rate (m^3/s)	Maximum daily volume (m^3/d)
NN 80955045	Intake	0.65	56160

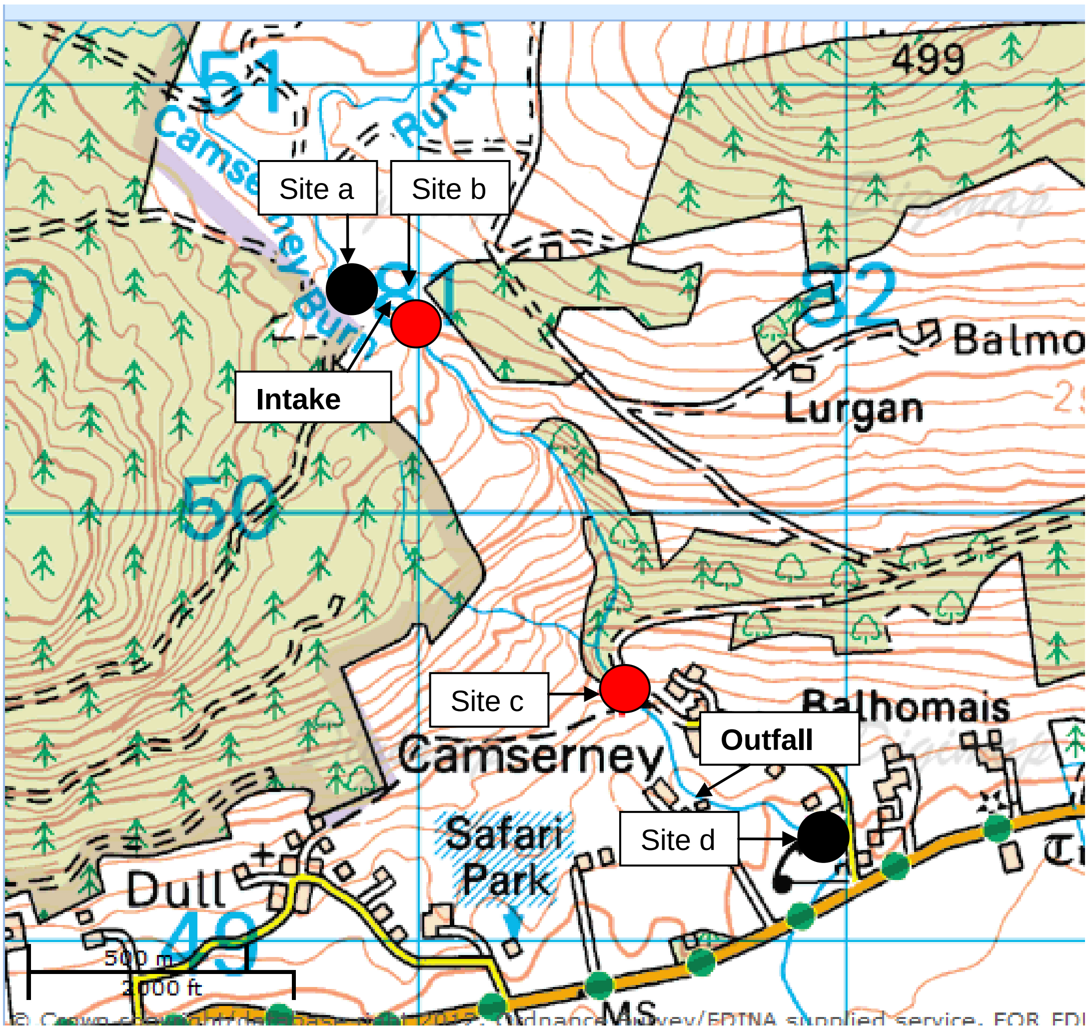

Figure 3.16 Location of intake, outfall and survey sites on Camserney Burn (red dots indicate impact sites located within the depleted stretch, black dots indicate a control site in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of the intake and one downstream of the outfall).

(a)

Figure 3.17 Intake structure at Camserney Burn.

3.2.10 Allt Gleann Da-Eig

The Allt Gleann Da-Eig run-of-river hydropower scheme, located in Glen Lyon, Scotland's longest glen, is estimated to have a generating capacity of 807 kW. The hydropower scheme was due for commissioning in spring 2011 but a number of construction delays meant the scheme did not become operational until October 2011. The upper reaches of the Allt Gleann Da-Eig flow through moor land before descending through a steep gorge to the valley floor joining the confluence of the River Lyon, a tributary of the River Tay. The Allt Gleann Da-Eig hydropower scheme has an intake at NN 61034611 and outfall at NN 61214734 approximately 150 m upstream of the confluence with the River Lyon (Table 3.20; Figure 3.18).

Fisheries surveys were carried out in 2007, 2008 and 2010 prior to commencement of construction of the scheme and in 2011 following construction of the scheme but prior to abstraction which commenced on 6th October 2011. Although the scheme did not commence abstraction until after the conclusion of the study fish surveys, the data were included to identify if there was any impact of the hydropower scheme construction.

Table 3.20 Fisheries survey site details in Allt Gleann Da-Eig in September 2006-2008 and 2010-2011 (BP = Backpack, G = Generator, Q = Quantitative, SQ = Semi-quantitative, HIFI – University of Hull International Fisheries Institute). Site identifiers marked with * indicate location within abstracted reach.

Site identifier/NGR	Site name	Survey dates	Length/mean width/area	Survey gear/method/ team
a NN60834597	Upstream intake, upstream of first bend, upstream of wall where river straightens	11/09/11 18/09/10 17/09/08 16/09/07	40 m/6.3 m/252 m^2 40 m/6.1 m/244 m^2 50 m/6.6 m/330 m^2 50 m/7.0 m/350 m^2	BP / Q / HIFI BP / Q / HIFI BP / SQ / HIFI BP / SQ / HIFI
b* NN61154632	Downstream of 1st left hand bend in river, downstream intake	11/09/11 18/09/10 17/09/08 16/09/07	40 m/9.2 m/368 m^2 40 m/5.8 m/232 m^2 50 m/8.0 m/400 m^2 50 m/3.5 m/175 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
c* NN61224725	Upstream of road bridge, upstream of outfall	11/09/11 18/09/10 17/09/08	30 m/9.9 m/297 m^2 30 m/8.5 m/255 m^2 30 m/5.0 m/150 m^2	BP / SQ / HIFI BP / SQ / HIFI BP / SQ / HIFI
d* NN61224730	Upstream of outfall, immediately downstream of road bridge	14/09/11 18/09/10 17/09/08 17/09/07	49 m/6.3 m/309 m^2 53 m/5.2 m/276 m^2 54 m/6.9 m/372 m^2 55 m/4.8 m/264 m^2	BP / Q / HIFI BP / Q / HIFI G / Q / HIFI G / SQ / HIFI
e NN61214740	Downstream outfall, downstream of split channel	11/09/11 18/09/10 17/09/08 17/09/07	40 m/8.2 m/328 m^2 42 m/6.7 m/281 m^2 49 m/8.4 m/412 m^2 38 m/6.9 m/262 m^2	BP / Q / HIFI BP / Q / HIFI G / Q / HIFI G / Q / HIFI

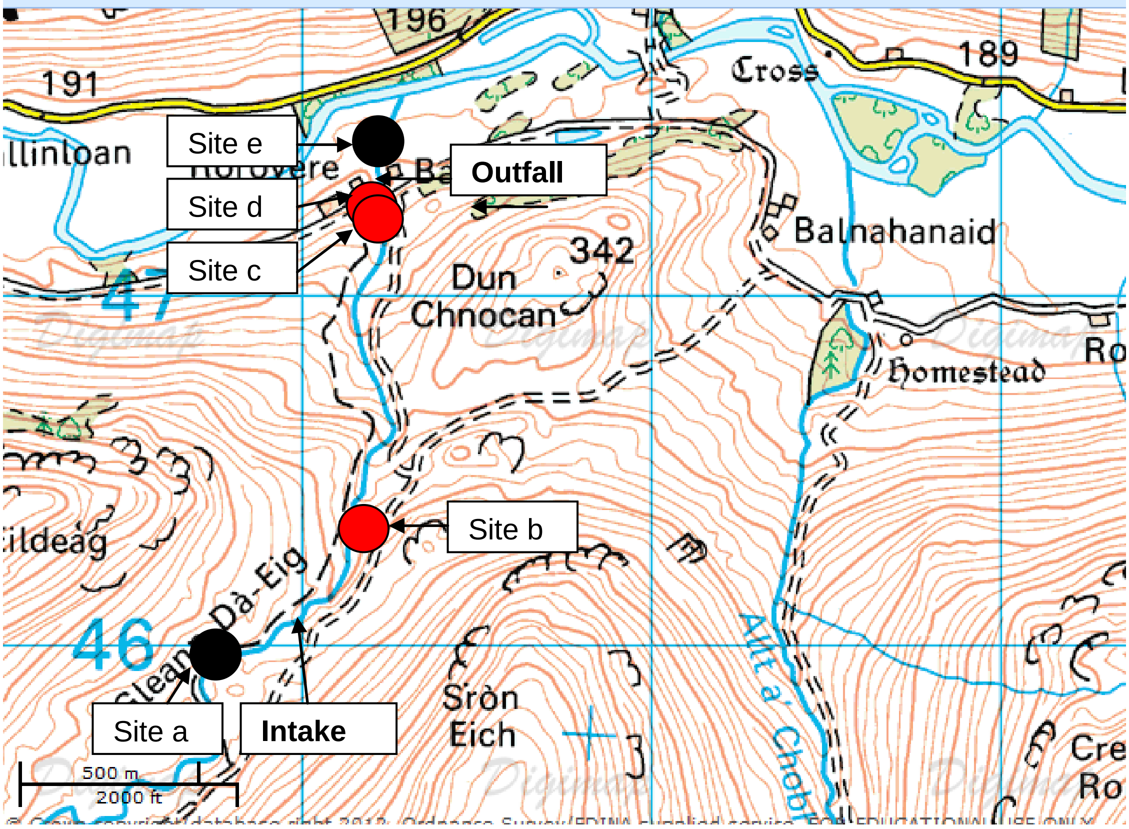

Figure 3.18 Location of intake, outfall and survey sites on Allt Gleann Da-Eig (red dots indicate impact sites located within the depleted stretch, black dots indicate control sites in both the upper and lower reaches situated outside of the depleted stretch, one site upstream of the intake and one downstream of the outfall).

An off-take weir was built at the intake and the intake and outfall structures were guarded by coanda screens to prevent entrainment of juvenile salmonids (Figure 3.19). It should be noted that the intake structure differs from many of the other schemes under assessment as the Allt Gleann Da-Eig scheme utilises a coanda screen at the intake traversing the whole river width, whereas other schemes utilise screens set alongside the river channel.

(a) (b)

Figure 3.19 Intake (photo a) and outfall (photo b) structure at Allt Gleann Da-Eig.

Modelled flow data indicated the scheme would divert an increasingly greater proportion of the water above Q_{85}, which increases to a maximum at about Q_{10}. The proportion of flow diverted progressively reduces as the flow regime approaches Q_{85} at which point abstraction ceases. Rates of abstraction are detailed in Table 3.21.

Table 3.21 Rates of abstraction at Allt Gleann Da-Eig.

NGR	Intake reference	Maximum abstraction rate (m^3/s)	Maximum daily volume (m^3/d)	Hands-off Flow (m^3/s)
NN 61034611	Intake	0.63	54432	0.067

3.3 Fisheries surveys and data collection

Electric fishing surveys were carried out in the same month each year to allow comparison of data. As salmonids hatch in spring, surveying was carried out during September, by this time of year young-of-the-year fish are of a larger size and thus more vulnerable to electric fishing, providing a more accurate and reliable account of the population. For each hydropower scheme surveyed, a minimum of four sites was sampled accounting for two control sites, one upstream of the intake and one downstream of the outfall, and two impact sites, one downstream of the intake and one upstream of the outfall. On several occasions the number of sites sampled was greater than the minimum requirement (Section 3.4). For schemes that had multiple intakes the number of sites sampled was greater as one extra control and impact site was surveyed on the additional intake(s). On schemes where the length of river between the intake and outfall was considerably long, additional sites within the impact reach were surveyed to provide reliable measures of both spatial and temporal changes in abundance of the species. To ensure good representation of the population diversity and structure were gained, each sample site was of sufficient length to incorporate complete sets of the characteristic habitat types including riffles, runs and pools.

Surveys involved a combination of quantitative and semi-quantitative electric fishing. Quantitative surveys involved three runs of electric fishing with the method aiming for a depletion between each run; this allowed for three-catch removal methods using both Zippin (1956) and Carle & Strub (1978) to produce estimates of absolute abundance. Semi-quantitative surveys were based on just a single run of electric fishing but population estimates were subsequently derived from calibration of the equipment (see Section 3.3.2); highlighting the importance of at least one quantitative survey on each river.

The type of survey gear used was dependent on accessibility to sites and the survey team undertaking the work (i.e. HIFI, SEPA, MFC, or LFT), but was either back pack electric fishing gear or bank-based equipment consisting of a generator powering a control box. Due to the nature of sites located in the upstream reaches, sites associated with the intake were always surveyed using back pack electric fishing gear. Sites associated with the outfall were generally surveyed using bank-based equipment; back pack electric fishing gear was used occasionally when accessibility to a site was limited. Section 3.4 presents the electric fishing method and equipment type used in each survey. During quantitative surveys, stop nets were used to prohibit fish movement in and out of the survey area; when site locations were difficult to access or excessive distances from vehicle access, natural obstacles such as small waterfalls and cascades provided barriers to fish movement. During the fishing exercise as many fish as possible were caught. Fishing in an upstream direction, two people carried dip nets, positioned either side and slightly downstream of a third person positioned in the middle with the anode.

After each of the three runs during quantitative surveys, fish were placed into one of three large buckets on the bank side; each bucket assigned to the relevant run number. After all runs were completed, fish caught from each run were identified to species according to Maitland (2004), measured to the nearest mm (fork length) and scales were taken from five fish within every 10 mm length class. Scales were only taken from fish >50 mm; fish smaller than this may not yet have laid down their first scales and thus it is presumed with great confidence that these fish would be young-of-the-year. Fish were returned immediately to the river at the site they were caught. The same procedure was performed after the first and only run of semi-quantitative surveys. At all sites, habitat data were collected and recorded on Scottish Fisheries Co-ordination Centre (SFCC) survey sheets for later analysis. At the key salmonid sites (where large numbers of salmon and trout were caught), mainly located in the lower reaches, additional HABSCORE data were collected for measuring and evaluating stream salmonid habitat features (Wyatt *et al.*, 1995) (see Section 3.4.5).

3.4 Data analysis

3.4.1 Length distributions

Fish lengths recorded in the field at each site allowed length distributions of salmon, trout and eels to be derived where sufficient numbers of fish were caught. For each species, each fish length was assigned into a 5 mm length class and the total number of fish within each length class was determined. Length distributions of trout and salmon, combined with ageing of scales, separated the 0+ fish from all other ages, classed as ≥1+. Please note that length frequency histograms were not produced when small numbers of fish were caught.

Length measurements or growth data were not available for sites surveyed by MFC and LFT; scale samples for growth analysis were not available from sites surveyed by SEPA, MFC or LFT. In hydropower schemes surveyed by HIFI, numbers and total lengths (mm) of eels were recorded. At sites surveyed by MFC, eels were recorded in abundance categories (1-10), while at sites surveyed by SEPA in 2011 eels <334 mm were measured and larger eels counted.

3.4.2 Density estimates

Density estimates of 0+ and ≥1+ fish/100 m^2 (calculated separately for each species) at quantitative sites were derived from absolute abundance estimates determined from two removal methods, Zippin (1956) and Carle & Strub (1978). Density figures were produced from density estimates using the Zippin method as recommended by Scottish Fisheries Co-ordination Centre (SFCC). In some cases, however, the Zippin (1956) method did not produce a density estimate as the number of fish caught between each run did not decline, as such there was no depletion; under these circumstances the density estimates were calculated using Carle & Strub (1978) method. In all cases the population density at each site was expressed as N/100 m^2. At quantitative sites the efficiency of sampling effort or probability of capture (*P*) was calculated from the Maximum Likelihood Methods (Zippin and Carle & Strub) and was used to calibrate the survey gears.

At semi-quantitative sites, surveyed by HIFI, for the derivation of density estimates, the method of gear calibration was used. This uses the probability of capture (*P*) derived from surveys with the same equipment at adjacent quantitative sites, and relative density (N/100 m^2) was derived as $N = ((C / P) / A) \times 100$, where C is the total number of fish caught in the single run and A is the sampling area (Cowx, 1996). It should be noted that the gear calibration method was not used for sites surveyed by SEPA, MFC and LFT as the quantitative sites were often not in the similar reach as the semi-quantitative sites. Density estimates of eels were not calculated as catches in the second and third runs were often greater than earlier runs; this contradicts one of the main assumptions of depletion sampling, namely that the population is reduced on each sampling run.

Statistical analysis was performed on density estimates from sites located within the ‘impacted’ reach to compare mean density before and after hydropower commissioning. Data were checked for homogeneity of variance using a Levene’s test.

If the variances were equal a paired two sample assuming equal variances *t*-test was performed, if variances were not equal a paired two sample assuming unequal variances *t*-test was performed.

3.4.3 Classification of population estimates

Density estimates were used to assess the status of salmon and trout populations and compared to five current fisheries classification schemes (FCS) used in UK fisheries assessment;

- SFCC-FCS – National classification scheme, using first run density estimates
- SFCC-FCS – National classification scheme taking account of influence of river width, using first run density estimates
- SFCC-FCS – Regional classification scheme, west and east, using first run density estimates
- SFCC-FCS – Regional classification scheme taking account of influence of river width, west and east, using first run density estimates
- EA-FCS – National classification scheme, using triple run data (or calibrated single run data)

For the four SFCC-FCS schemes, density estimates were derived from the first run only according to the scheme developed by Godfrey (2005), however, for the EA-FCS quantitative density estimates from the Carle & Strub method were used and at semi-quantitative sites the gear calibration method was used to provide a density estimate.

To provide an indication of the status of salmonid populations, density estimates were awarded classifications based on a grading scale (A-F) translated as follows: Grade A (excellent), Grade B (good), Grade C (fair or average), Grade D (fair/poor), Grade E (poor) and Grade F (fishless). The EA-FCS was developed to allow comparison of juvenile salmon monitoring data with a juvenile database derived from over 600 survey sites in England and Wales (Mainstone *et al.,* 1994). The SFCC-FCS was developed and based on the EA scheme but used a database of survey sites in Scotland (Godfrey, 2005). Whilst the same grading system applies (A-F) there is no formal naming of the grade classes of the SFCC-FCS and so the same naming convention is followed for the purpose of this study. It should be noted that the SFCC–FCS grading system is based on single run electric fishing surveys, hence at all survey sites the first run catch was used for classification of the salmonid populations, whereas for the EA-FCS triple run data or calibrated single run data were used.

When classifying the salmonid populations based on the regional SFCC classification scheme all rivers were assigned to the east regional classification scheme using the areas defined by Godfrey (2005) with the exception of the River Callop and Douglas Water hydropower schemes which were assigned to the west regional classification scheme. The Rottal Burn hydropower scheme, located in the South Esk, was located in a region that was not covered by the work of Godfrey (2005), therefore the river was assigned to the east regional classification as this was the nearest adjacent region. The population density grades for all classification schemes are detailed in Tables 3.22-3.27.

Table 3.22 Salmonid abundance (N/100 m^2) classifications used in the SFCC-FCS – National classification scheme (adapted from Godfrey, 2005). Note salmonid abundance classifications based on single run electric fishing data.

	Abundance classification					
Species group	A	B	C	D	E	F
0+ brown trout	≥30.3	12.4-30.2	5.3-12.3	2.5-5.2	0.1-2.4	0
≥1+ brown trout	≥10.4	5.6-10.3	3.1-5.5	1.6-3.0	0.1-1.5	0
0+ salmon	≥42.1	20.3-42.0	10.3-20.2	4.7-10.2	0.1-4.6	0
≥1+ salmon	≥15.8	9.1-15.7	5.1-9.0	2.6-5.0	0.1-2.5	0

Table 3.23 Salmonid abundance (N/100 m^2) classifications used in the SFCC-FCS – National classification scheme taking account of influence of river width (adapted from Godfrey, 2005). Note salmonid abundance classifications based on single run electric fishing data.

	Abundance classification					
Species group/river width	A	B	C	D	E	F
<4 m						
0+ brown trout	≥49.9	22.9-49.8	11.0-22.8	4.5-10.9	0.1-4.4	0
≥1+ brown trout	≥15.3	8.3-15.2	5.0-8.2	4.5-4.9	0.1-4.4	0
0+ salmon	≥35.2	15.2-35.1	8.7-15.1	4.3-8.6	0.1-4.2	0
≥1+ salmon	≥15.8	8.3-15.7	5.1-8.2	2.5-5.0	0.1-2.4	0
4-6 m						
0+ brown trout	≥32.2	13.7-32.1	6.9-13.6	3.3-6.8	0.1-3.2	0
≥1+ brown trout	≥10.3	6.4-10.2	3.4-6.3	2.0-3.3	0.1-1.9	0
0+ salmon	≥49.2	26.6-49.1	11.0-26.5	5.1-10.9	0.1-5.0	0
≥1+ salmon	≥16.8	9.6-16.7	5.1-9.5	2.8-5.0	0.1-2.7	0
6-9 m						
0+ brown trout	≥12.9	5.7-12.8	4.0-5.6	2.2-3.9	0.1-2.1	0
≥1+ brown trout	≥6.2	3.6-6.1	2.1-3.5	1.3-2.0	0.1-1.2	0
0+ salmon	≥41.2	21.5-41.1	12.2-21.4	5.2-12.1	0.1-5.1	0
≥1+ salmon	≥16.8	10.6-16.7	6.2-10.5	3.2-6.1	0.1-3.1	0
>9 m						
0+ brown trout	≥7.1	3.3-7.0	1.8-3.2	1.1-1.7	0.1-1.0	0
≥1+ brown trout	≥2.7	1.8-2.6	1.0-1.7	0.7-0.9	0.1-0.6	0
0+ salmon	≥38.9	18.7-38.8	10.7-18.6	4.2-10.6	0.1-4.1	0
≥1+ salmon	≥14.2	8.2-14.1	4.1-8.1	2.3-4.0	0.1-2.2	0

Table 3.24 Salmonid abundance (N/100 m^2) classifications used in the SFCC-FCS – Regional classification scheme (adapted from Godfrey, 2005). Note salmonid abundance classifications based on single run electric fishing data.

Species group/area	Abundance classification A	B	C	D	E	F
East						
0+ brown trout	≥72.10	26.21-72.09	11.94-26.20	4.31-11.93	0.1-4.30	0
≥1+brown trout	≥13.85	7.46-13.84	3.39-7.45	1.86-3.38	0.1-1.85	0
0+ salmon	≥104.58	43.38-104.57	21.54-43.37	6.89-21.53	0.1-6.88	0
≥1+ salmon	≥19.07	10.16-19.06	6.30-10.15	3.05-6.29	0.1-3.04	0
West						
0+ brown trout	≥50.40	17.29-50.39	5.53-17.28	1.99-5.52	0.1-1.98	0
≥1+brown trout	≥8.36	5.28-8.35	3.05-5.27	1.59-3.04	0.1-1.58	0
0+ salmon	≥26.58	14.59-26.57	6.05-14.58	2.44-6.04	0.1-2.43	0
≥1+ salmon	≥11.27	5.81-11.26	3.64-5.80	1.93-3.63	0.1-1.92	0

Table 3.25 Salmonid abundance (N/100 m^2) classifications used in the SFCC-FCS – Regional classification scheme (West) taking account of influence of river width (adapted from Godfrey, 2005). Note salmonid abundance classifications based on single run electric fishing data.

Species group/width	Abundance classification A	B	C	D	E	F
<4 m						
0+ brown trout	>74.4	44.7-74.3	28.5-44.6	9.9-28.4	0.1-9.8	0
≥1+ brown trout	>12.1	7.6-12.0	5.6-7.5	3.9-5.5	0.1-3.8	0
0+ salmon	>17.2	10.7-17.1	5.3-10.6	2.4-5.2	0.1-2.3	0
≥1+ salmon	>12.2	6.9-12.1	3.3-6.8	2.3-3.2	0.1-2.2	0
4-6 m						
0+ brown trout	>19.0	12.4-18.9	5.0-12.3	3.0-4.9	0.1-2.9	0
≥1+ brown trout	>8.4	5.4-8.3	3.3-5.3	2.3-3.2	0.1-2.2	0
0+ salmon	>35.5	14.0-35.4	6.0-13.9	3.5-5.9	0.1-3.4	0
≥1+ salmon	>10.8	6.6-10.7	5.0-6.5	2.0-4.9	0.1-1.9	0
6-9 m						
0+ brown trout	>5.3	2.7-5.2	1.8-2.6	1.1-1.7	0.1-1.0	0
≥1+ brown trout	>4.9	3.2-4.8	2.1-3.1	1.5-2.0	0.1-1.4	0
0+ salmon	>21.1	14.0-21.0	10.4-13.9	1.6-10.3	0.1-1.5	0
≥1+ salmon	>10.9	5.9-10.8	4.4-5.8	1.9-4.3	0.1-1.8	0
>9 m						
0+ brown trout	>4.0	2.6-3.9	1.5-2.5	0.8-1.4	0.1-0.7	0
≥1+ brown trout	>1.8	1.5-1.7	0.9-1.4	0.7-0.8	0.1-0.6	0
0+ salmon	>45.1	15.9-45.0	8.1-15.8	2.7-8.0	0.1-2.6	0
≥1+ salmon	>6.6	4.2-6.5	3.2-4.1	1.7-3.1	0.1-1.6	0

Table 3.26 Salmonid abundance (N/100 m^2) classifications used in the SFCC-FCS – Regional classification scheme (East) taking account of influence of river width (adapted from Godfrey, 2005). Note salmonid abundance classifications based on single run electric fishing data.

Species group/width	Abundance classification A	B	C	D	E	F
<4 m						
0+ brown trout	>108.9	70.2-108.8	32.9-70.1	17.4-32.8	0.1-17.3	0
≥1+ brown trout	>20.4	14.1-20.3	8.9-14.0	4.4-8.8	0.1-4.3	0
0+ salmon	>124.0	53.8-123.9	20.5-53.7	5.5-20.4	0.1-5.4	0
≥1+ salmon	>20.3	10.6-20.2	6.3-10.5	3.0-6.2	0.1-2.9	0
4-6 m						
0+ brown trout	>89.7	42.6-89.6	16.8-42.5	9.0-16.7	0.1-8.9	0
≥1+ brown trout	>13.2	7.5-13.1	4.7-7.4	2.7-4.6	0.1-2.6	0
0+ salmon	>63.6	37.2-63.5	21.5-37.1	7.1-21.4	0.1-7.0	0
≥1+ salmon	>16.7	8.9-16.6	6.3-8.8	3.2-6.2	0.1-3.1	0
6-9 m						
0+ brown trout	>26.5	13.2-26.4	6.6-13.1	4.1-6.5	0.1-4.0	0
≥1+ brown trout	>9.4	3.7-9.3	2.9-3.6	1.8-2.8	0.1-1.7	0
0+ salmon	>90.9	45.2-90.8	22.5-45.1	12.5-22.4	0.1-12.4	0
≥1+ salmon	>22.0	12.6-21.9	8.1-12.5	4.1-8.0	0.1-4.0	0
>9 m						
0+ brown trout	>12.2	6.1-12.1	3.9-6.0	1.7-3.8	0.1-1.6	0
≥1+ brown trout	>3.0	1.6-2.9	0.8-1.5	0.5-0.7	0.1-0.4	0
0+ salmon	>79.8	37.3-79.7	17.1-37.2	5.6-17.0	0.1-5.5	0
≥1+ salmon	>15.5	9.9-15.4	5.3-9.8	2.3-5.2	0.1-2.2	0

Table 3.27 Salmonid abundance (N/100 m^2) classifications used in the EA-FCS. Note salmonid abundance classifications based on triple run data or calibrated single run data.

Species group	Abundance classification A	B	C	D	E	F
0+ brown trout	≥38.0	17.0-37.9	8.0-16.9	3.0-7.9	0.1-2.9	0
≥1+ brown trout	≥21.0	12.0-20.9	5.0-11.9	2.0-4.9	0.1-1.9	0
0+ salmon	≥86.0	45.0-85.9	23.0-44.9	9.0-22.9	0.1-8.9	0
≥1+ salmon	≥19.0	10.0-18.9	5.0-9.9	3.0-4.9	0.1-2.9	0

3.4.4 Growth rates

The determination of the age and growth of fish is an important tool in the assessment of fish population dynamics (Bagenal, 1978); scale ageing was the basis for indicating growth rates of salmon and trout. When large numbers of scale samples were collected, sub-sampling of a representative number was carried out according to the Environment Agency Management System (Britton, 2003). Samples of scales were taken from five fish within each 10 mm length class; three of these samples were aged. The age and growth of salmon and trout were determined in each river by the interpretation and counting of annual growth checks – referred to as annuli; these appear on the scales of the fish (Bagenal & Tesch, 1978). Annuli are formed between periods of faster and little or no growth, the latter generally occurring during the winter months in temperate regions. During summer when food is plentiful, faster growth is represented by widely-spaced ridges (circuli), during times of slow growth, often over winter the rings are much closer together and may overlap. Each section of slow and fast growth is classed as 1 year.

Scales from individual fish were examined under a microfiche projector and the fish were aged by counting the number of annuli, taking care to note any false checks. False checks are generally formed by stress and should not be counted. More than one individual scale was examined to ensure correct interpretation of the annuli. The total scale radius and the scale radius to each annulus were measured from the nucleus. Analysis of the data involved assessment of the following relationship between the length of the fish, scale radius to annuli and total scale radius (Dahl-Lea method; Francis, 1990):

$L_i = (S_i/S_c) \times L_c$

where L_i is length (mm) at year i, S_i scale radius at length L_i, L_c length at capture and S_c scale radius at capture. For each individual fish, the length at age was calculated from the scale radius to each annulus at each age. This calculation was repeated for each fish and the mean lengths for each age from all fish were calculated. Data were then tabulated for salmon and trout at each survey site. Growth rates of salmon and trout at sites in close proximity were derived from combined data.

During the surveys in 2011, 0+ fish would not yet have reached age 1 therefore the back-calculated length for age 1 could not be calculated in this instance. Instead, mean observed length was calculated from 0+ individuals caught at sites in 2011. Surveys were carried out in September (each year) which is towards the end of the growth year; as such the mean observed length of 0+ individuals caught in the 2011 surveys was considered a reliable indicator of first year growth; this allowed direct comparison to the back-calculated length at age 1 in other survey years.

3.4.5 HABSCORE data collection and outputs

HABSCORE is a system for measuring and evaluating stream salmonid habitat features based on empirical statistical models relating the population size of five salmonid species/age combinations (Wyatt *et al.,* 1995). The method is a predictive tool commonly used in the assessment of stream habitat features statistically linked to population estimates of salmon and trout. The model was designed, tested and refined for use in England and Wales; the applicability of HABSCORE to Scottish rivers has not been quantified but the outputs still provide a valuable comparison of observed against expected abundances. Using the information from three HABSCORE questionnaires, the software produces a series of outputs, which includes estimates of the expected populations (the Habitat Quality Score, HQS) and the degree of habitat utilisation (the Habitat Utilisation Index, HUI), for each of five salmonid species/age combinations (Wyatt *et al.,* 1995). HABSCORE outputs were used to identify variation in the observed densities, predicted densities and habitat utilisation of 0+ salmon, ≥1+ salmon, 0+ trout, ≥1+ (<20 cm) trout and ≥1+ (>20 cm) trout prior to and following commissioning of hydropower schemes. Pre- and -post-hydropower scheme commissioning HABSCORE data were only available for rivers surveyed by HIFI (except Camserney Burn) as SEPA, MFC and LFT did not collect HABSCORE data. HABSCORE was not carried out at Camserney Burn as there were no pre-commissioning data.

To collect information for HABSCORE analysis a questionnaire on the habitat found at key salmonid sites was completed following each fisheries survey. Starting at the bottom of the survey site the wetted width of the river was measured and depth measurements were taken at three equally spaced intervals corresponding to ¼, ½ and ¾ along this measured wetted channel width. The first 10 m section walking upstream was measured and the wetted width and the cross channel depth profile at the top of the first section were recorded. Within the 10 m section the substrate composition (bedrock/artificial, boulders, cobbles, gravel/coarse sand, fine sand and compacted clay) and flow type (cascade/torrential, turbulent/broken deep, turbulent/broken shallow, glide/run deep, glide/run shallow, slack deep and slack shallow) were recorded based on the percentage of five abundance categories; dominant, frequent, common, scarce and absent. These procedures were repeated every 10 m upstream until the top of the survey site was reached. If the last section of river was <15 m the exact length was noted and treated as one section, if the length was >15 m it was split into another two sections with the first being a 10 m section and the last section the remaining length; for example 16 m would be treated as a 10 m section and a 6 m section. Information collected would allow for later evaluation of salmonid stream habitat features.

The methodology of habitat data collection and completion of the relevant form (HABform), along with completion of river catchment information (MAPform) and fisheries information (FISHform), are documented by Barnard & Wyatt (1995).

Data from the three completed forms (HABform, MAPform and FISHform) at each site were entered into the HABSCORE for Windows program and the following outputs were produced for salmon and trout populations (definitions from Wyatt *et al.,* 1995).

Habitat Quality Score (HQS)

The HQS value is a measure of the habitat quality expressed as the expected long-term average density of fish (N/100 m^2). The HQS is derived from habitat and catchment features, and assumes that neither water quality nor recruitment are limiting the populations. The HQS is used as an indicator of the potential of the site, against which the observed size of populations may be compared.

HQS lower and upper confidence limits

These are the lower and upper 90% confidence limits for the HQS, N/100 m^2. The confidence limits given should enclose the average observed density for a site on 90% of occasions. The probability of getting an observed average density lower than the lower confidence limit by chance alone is therefore 5%.

Habitat Utilisation Index (HUI)

The HUI is a measure of the extent to which the habitat is utilised by salmonids. It is based on the difference between the 'observed' density and that which would be expected under 'pristine' conditions (i.e. the HQS). When the 'observed' density and the HQS are identical, the HUI takes the value of one; HUI values less than one will occur when the observed densities are less than expected.

HUI lower and upper confidence limits

These are the upper and lower 90% confidence limits for the HUI, expressed as a proportion. An upper HUI confidence interval <1 indicates that the observed population was significantly less than would be expected under pristine conditions. Conversely, a lower HUI confidence interval >1 indicates that the observed population was significantly higher than would normally be expected under pristine conditions.

Log_e HUI

This is the natural logarithm of the HUI. Negative values will represent an observed population less than that which would be expected given the habitat. The data were tabulated from each site and interpreted in relation to the fish population data.

3.4.6 Wetted widths

Individual wetted widths collected during HABSCORE surveys both pre-and post-hydropower commissioning were tested for homogeneity of variance (Bartlett's test) and normal distribution (two-sample Kolmogorov-Smirnov). When data conformed to normal distribution and homogeneity of variance a paired two sample *t*-test was performed. In all other instances a Mann Whitney *U* test was performed. Mean wetted width before (all years combined) and after (all years combined) the scheme was commissioned were compared at a significance level of 0.05. This analysis was performed only at those sites located within the abstracted reach of the river. The purpose was to check there was no significant difference in the wetted widths between HABSCORE surveys over the study period. This would allow confirmation that surveys were carried out at a comparable river width and thus confirm density estimates over the study period were comparable. It should be noted that the term comparable is not inferring that density estimates were the same before and after the scheme was commissioned. It is used in reference to the fact that density estimates were comparable because there had not been a significant reduction in the wetted width causing an ‘artificial increase’ in density estimates after hydropower commissioning.

As SEPA, MFC and LFT did not collect HABSCORE data, wetted width analysis could not be performed for the River Callop, Ardvorlich Burn and Douglas Water.

CHAPTER FOUR

4 IMPACTS OF RUN-OF-RIVER HYDROPOWER SCHEMES ON FISH POPULATIONS: STAGE ONE

4.1 Introduction

This chapter compares density, age structure and growth of Atlantic salmon and brown trout, before and after five of the ten run-of-river hydropower schemes under study were commissioned, as described in Chapter 3. These five schemes are Kinnaird Burn, Keltney Burn, Rottal Burn, Innerhadden Burn and Inverhaggernie Burn. Additionally, wetted widths and density of fish in relation to the habitat available Based on the monitoring programme undertaken, each scheme presents strong data sets allowing thorough analysis and interpretation. Analysis of data aims to isolate natural trends that are observed in fish populations from trends potentially in response to operation of the hydropower scheme.

4.2 Kinnaird Burn hydropower scheme

4.2.1 Salmonid population trends

No salmon was captured at any of the sites in Kinnaird Burn throughout the study period. The reach of the hydropower scheme was considered inaccessible to migratory salmonids due to flood damage to a fish pass in the lower reaches of the burn. Adult salmon have historically been found upstream of the fish pass (R. Gardiner, pers. comm.), but there were no historical survey data to suggest migratory salmonids spawn in the reach of the hydropower scheme.

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

Upstream of the intakes

0+ trout densities at site a, upstream of the main intake, varied between study years with the greatest density in 2009, classified as fair/average (class C), and the lowest density in 2010, classified as poor (class E) (Figure 4.1, Table 4.1). 0+ trout populations in 2008 and 2011 were similar and classed fair/poor (class D). ≥1+ trout densities at site a varied over the study period, from fair/average (class C) in 2008 to excellent (class A) in 2009 and 2010 (Figure 4.1, Table 4.1); the lowest densities were found in 2011 with populations classified as fair/poor (class D) (Figure 4.1, Table 4.1).

At site c, upstream of the intake on the small west tributary, 0+ trout were absent in 2008, but populations improved to good (class B) in 2010; 0+ trout densities were lower in 2011 and classed fair/poor (class D) (Figure 4.1, Table 4.1). ≥1+ trout populations at site c increased from fair/poor (class D) in 2008 to excellent (class A) in 2009 and 2010 decreasing to good (class B) in 2011; the highest density of ≥1+ trout occurred in 2010 (Figure 4.1, Table 4.1).

Table 4.1 Classification of trout densities in both the upper and lower reaches of Kinnaird Burn between 2005 and 2011, based on five classification schemes. Note hydropower scheme commissioned October 2008, n/s = not surveyed. Site identifiers marked with * indicate location within abstracted reach.

		0+ trout					≥1+ trout				
Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2005	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2006	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2007	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2008	D	E	E	E	D	C	C	C	D	C
	2009	C	D	D	E	D	A	B	B	C	B
	2010	E	E	E	E	E	A	A	A	B	A
	2011	D	E	E	E	D	D	D	E	E	D
b*	2005	D	E	E	E	D	B	B	B	B	B
	2006	E	E	E	E	D	A	B	B	C	B
	2007	C	D	D	E	C	A	A	A	B	A
	2008	E	E	E	E	D	A	A	A	A	A
	2009	B	C	B	D	B	B	A	A	B	A
	2010	C	D	D	E	C	A	A	A	B	A
	2011	E	E	E	E	E	C	C	E	E	D
c	2005	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2006	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2007	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2008	F	F	F	F	F	D	E	E	E	D
	2009	C	D	D	E	D	A	B	B	C	B
	2010	B	C	B	D	B	A	A	A	B	A
	2011	D	D	E	E	D	B	B	B	D	B
d*	2005	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2006	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2007	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s	n/s
	2008	C	C	C	E	B	B	B	C	D	C
	2009	F	F	F	F	F	A	A	A	A	A
	2010	B	C	C	D	B	A	A	A	A	A
	2011	E	E	E	E	E	B	B	B	C	C
e*	2005	D	E	E	E	D	A	A	B	B	B
	2006	C	D	C	E	C	A	B	A	B	B
	2007	D	E	E	E	D	A	A	A	B	A
	2008	C	D	D	E	C	A	A	A	B	B
	2009	C	D	D	E	C	A	A	A	A	A
	2010	E	E	E	E	E	A	A	A	B	A
	2011	F	F	F	F	F	D	D	E	E	D
f*	2005	C	D	D	E	C	A	B	B	C	A
	2006	C	C	C	D	B	B	C	B	C	C
	2007	C	D	D	E	C	A	A	A	A	A
	2008	E	E	E	E	E	A	A	A	B	A
	2009	B	C	C	D	B	A	B	B	C	A
	2010	D	D	D	E	D	A	A	B	B	B
	2011	E	E	E	E	E	B	B	B	C	B
g	2005	C	D	D	E	C	A	A	A	B	A
	2006	C	D	C	D	C	B	B	B	B	B
	2007	B	C	C	D	B	A	A	A	A	A
	2008	D	E	D	E	D	A	A	A	A	A
	2009	B	C	C	E	B	A	A	A	B	A
	2010	E	E	E	E	E	A	A	B	B	A
	2011	E	E	E	E	E	B	C	B	C	C

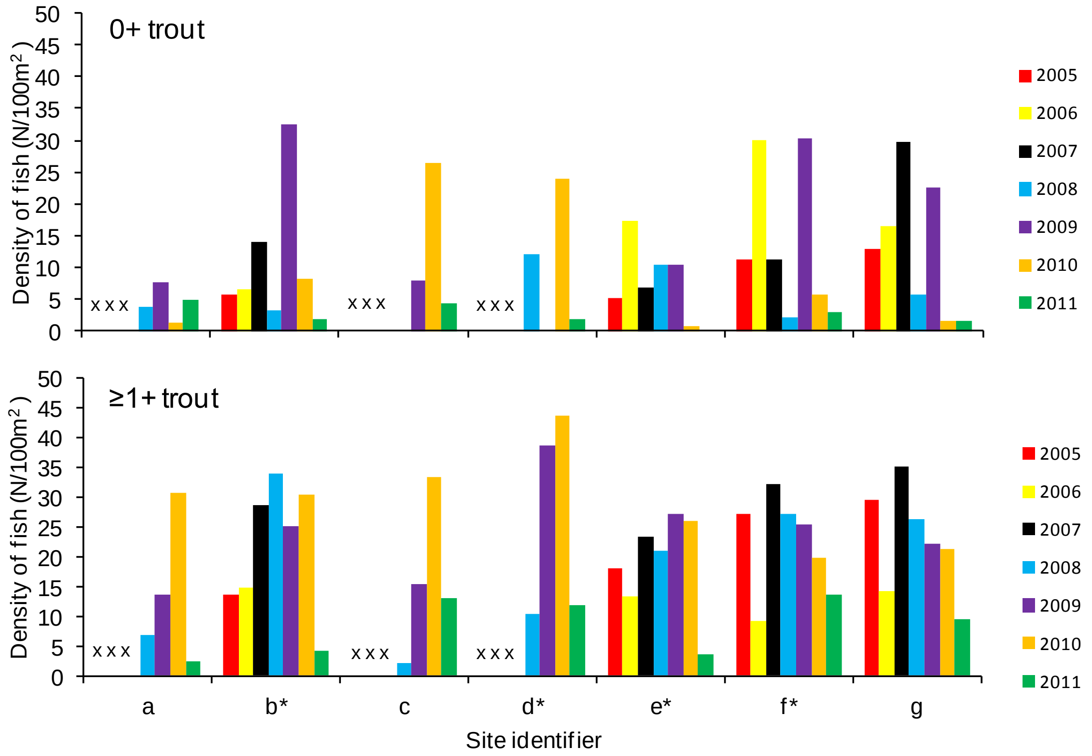

Figure 4.1 Density estimates of trout in Kinnaird Burn between 2005 and 2011. x represents years that were not sampled.

Depleted reach

At site b, downstream of the main intake, 0+ trout densities varied considerably over the study period with the greatest density in 2009 (Figure 4.1, Table 4.1). In 2005 and 2006, 0+ trout populations were classed fair/poor (class D) and poor (class E), respectively. 0+ trout populations increased in 2007 to fair/average (class C) but decreased in 2008 to poor (class E) before increasing considerably in 2009 to good (class B). In 2010 and 2011, 0+ trout populations decreased to fair/average (class C) and poor (class E), respectively, with the latter density similar to 2008 (Figure 4.1, Table 4.1). ≥1+ trout populations at site b were excellent (class A) in 2006, 2007, 2008, and 2010, with the greatest density occurring in 2008, and good (class B) in 2005 and 2009; ≥1+ trout populations were fair/average (class C) in 2011 (Figure 4.1, Table 4.1).

At site d, downstream of the intake on the small west tributary, 0+ trout populations varied between years with the greatest density in 2010 similar to findings at site c (Figure 4.1). 0+ trout densities were classed fair/average (class C) in 2008 increasing to good (class B) in 2010 before falling to poor (class E) in 2011; 0+ trout were absent in 2009 (Table 4.1). ≥1+ trout populations at site d were excellent (class A) in 2009 and 2010 and good (class B) in 2008 and 2011, with the greatest density in 2010 similar to findings at site c (Figure 4.1, Table 4.1).

At site e, located in the middle reaches of the abstracted reach, 0+ trout densities varied between years (Figure 4.1, Table 4.1). Between 2005 and 2009 0+ trout populations were either fair/average (class C) or fair/poor (class D), while in 2010 populations were poor (class E) and no 0+ trout were captured in 2011. ≥1+ trout populations generally increased between 2005 and 2010 and were excellent (class A) throughout, but populations were fair/poor (class D) in 2011 (Figure 4.1, Table 4.1).

Downstream of the outfall

At site f, upstream of the outfall, 0+ trout populations were classified as fair/average (class C) between 2005 and 2007 becoming poor (class E) in 2008 before improving to good (class B) in 2009; 0+ trout populations decreased in 2010 and 2011 being poor (class E) in the latter year and at a similar density as found in 2008 (Figure 4.1, Table 4.1). ≥1+ trout populations at site f were excellent (class A) throughout the study period except in 2006 and 2011 when densities were lower and populations were classified as good (class B) (Figure 4.1, Table 4.1).

At site g, downstream of the outfall, 0+ trout populations improved from fair/average (class C) in 2005 and 2006 to good (class B) in 2007; in 2008 0+ trout populations were fair/poor (class D) improving to good (class B) in 2009 before becoming poor (class E) in 2010 and 2011 (Figure 4.1, Table 4.1). These trends of poorer recruitment in 2010 and 2011 were also found at sites e and f within the abstracted reach as well as downstream of the abstracted reach at site g. ≥1+ trout populations at site g were excellent (class A) throughout the study period except in 2006 and 2011 when densities were lower and populations were classified as good (class B) (Figure 4.1, Table 4.1). The population trends of ≥1+ trout at site g, downstream of the abstracted reach, closely match those found at site f, within the abstracted reach (Figure 4.1).

4.2.2 Salmonid size structure

Upstream of the intake

Length distributions of brown trout in the upper reaches associated with the two intakes varied between years at each site. At site a and c, upstream of the intakes, 0+ trout were caught in small numbers throughout the study period, in the size range 50-70 mm (Figure 4.2) and 40-65 mm (Figure 4.4) respectively. At site a, ≥1+ trout in 2008 and 2011 were caught in a narrow size range, 115-140 mm and 140-175 mm, respectively, while in 2009 and 2010 ≥1+ trout were caught in a wider size range of 75-170 mm (Figure 4.2). ≥1+ trout at site c, were caught in a similar size range (85-155 mm) throughout the study period except in 2010 when one larger individual >180 mm was captured (Figure 4.4).

Depleted reach

At site b, downstream of the main intake, 0+ trout were caught in the size range 40-75 mm throughout the study period (Figure 4.3). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2005 and 2008 when a small number of larger individuals > 200 mm were captured (Figure 4.3); in 2011 ≥1+ trout were caught in a narrower size range of 100-145 mm reflecting the lower numbers present at the site and the size structure found at site a upstream of the main intake.

At site d, downstream of the intake on the small west tributary, 0+ trout were caught in a slightly narrower size range than at site c, namely 50-70 mm except in 2009 when 0+ trout were absent (Figure 4.5). ≥1+ trout were also caught in a slightly narrow size range (75-130 mm) than at site c during the study period (Figure 4.5).

In the middle reaches of Kinnaird Burn at site e 0+ trout were caught in a similar size range of 45-75 mm throughout the study period except in 2011 when 0+ trout were absent; only one 0+ trout in the 50-55 mm size range was caught in 2010 (Figure 4.6). ≥1+ trout were caught in a similar size range (80-200 mm) throughout the study period except in 2011 when fish were caught in a narrower size range of 105-200 mm, reflecting the size structure found at site g downstream of the outfall.

At site f, 0+ trout were caught in the size range 40-75 mm throughout the study period, albeit in small numbers in 2008, 2010 and 2011 (Figure 4.7). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2005, 2006, 2010 and 2011 when a small number of larger individuals > 180 mm were captured (Figure 4.7). In 2011 ≥1+ trout were caught in a narrower size range of 105-195 mm reflecting the lower numbers present at the site, and the size structure found at site g downstream of the outfall.

Downstream of the outfall

At site g, 0+ trout were caught in the size range 40-75 mm throughout the study period, albeit in small numbers in 2008, 2010 and 2011 (Figure 4.8). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2006, 2007, 2010 and 2011 when a small number of larger individuals > 180 mm were captured (Figure 4.8). In 2011 ≥1+ trout were caught in a narrower size range of 105-210 mm reflecting the lower numbers present at the site.

Overall during the study period the size structure of trout within the abstracted reaches followed similar trends to the size structure of trout at sites outside the abstracted reach.

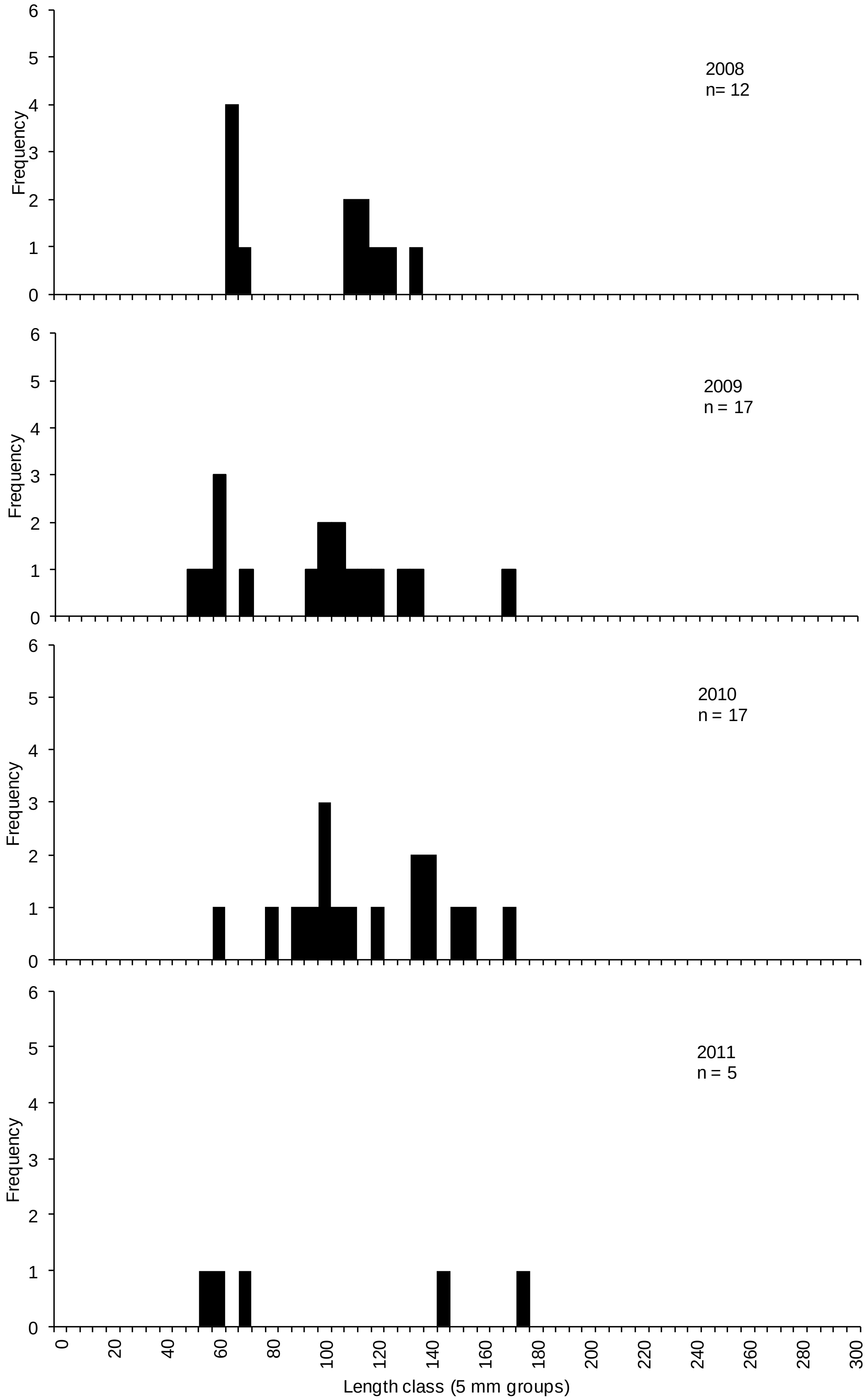

Figure 4.2 Length distributions of trout at site a in Kinnaird Burn between 2008 and 2011. Note site a was not surveyed between 2005-2007.

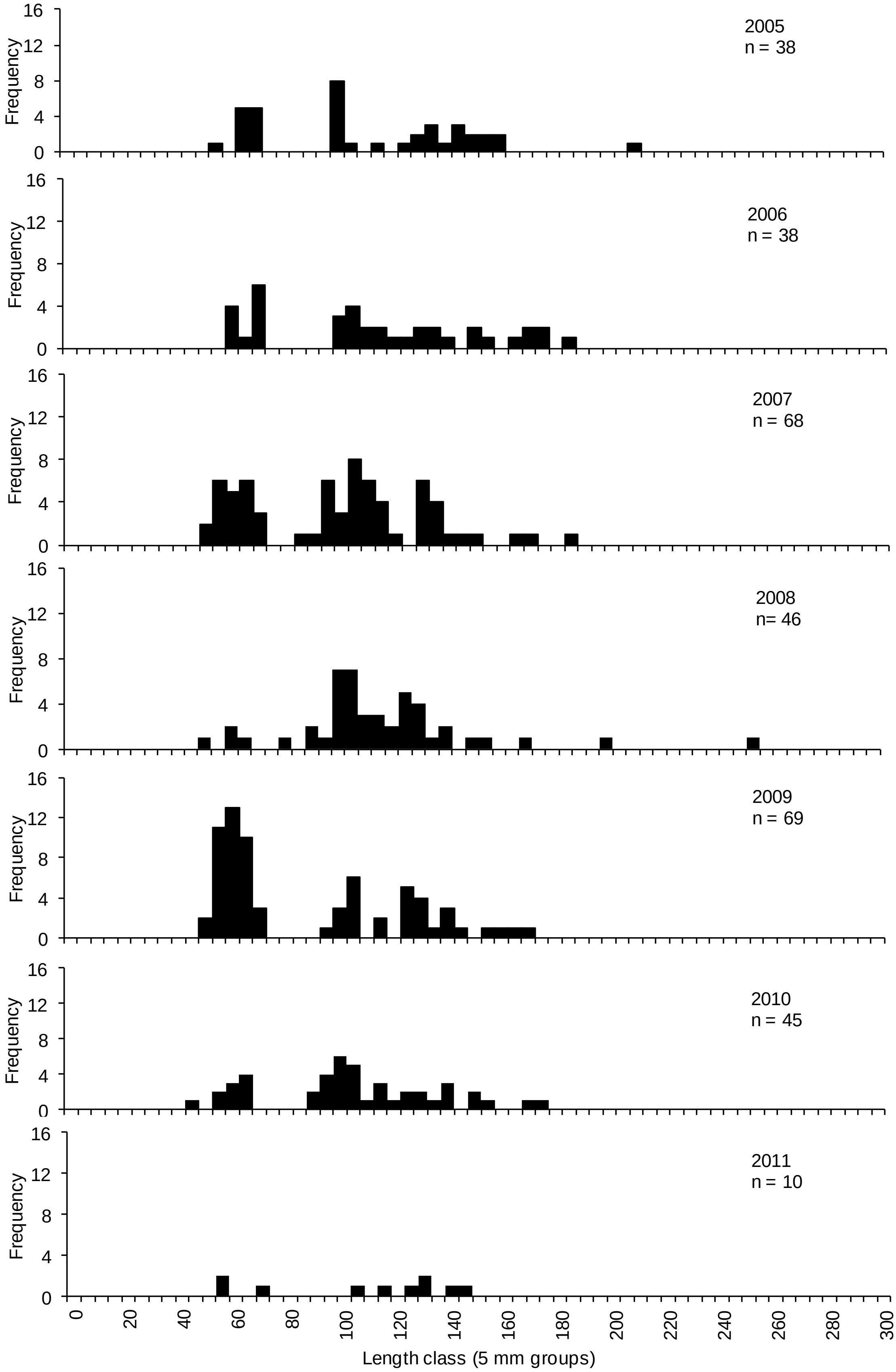

Figure 4.3 Length distributions of trout at site b in Kinnaird Burn between 2005 and 2011.

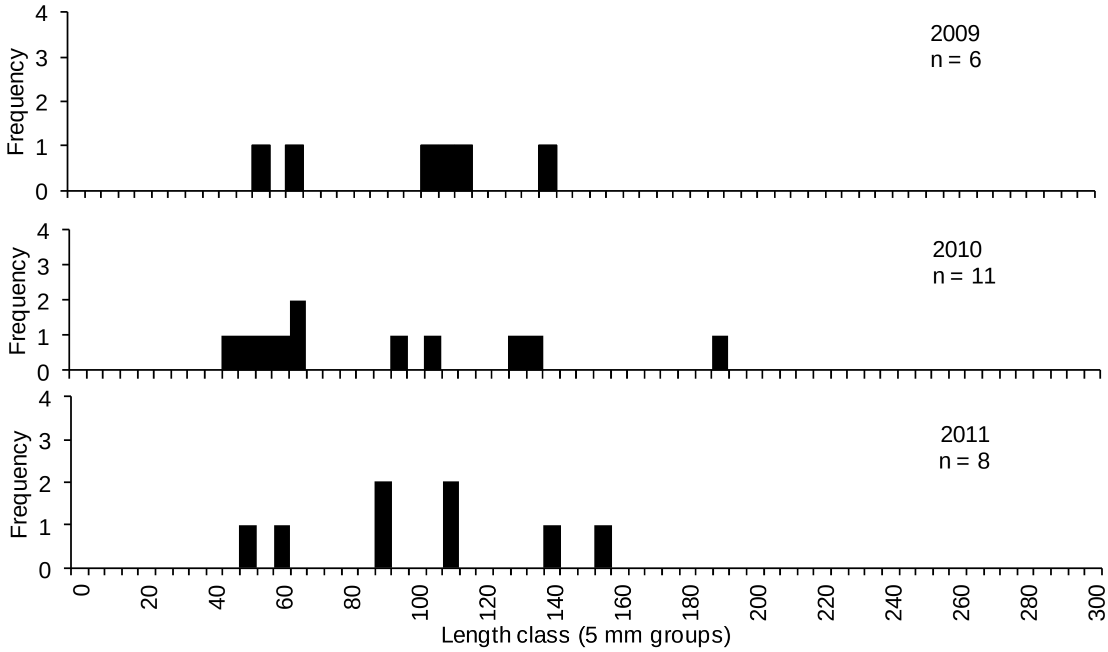

Figure 4.4 Length distributions of trout at site c in Kinnaird Burn between 2009 and 2011. Note one fish was captured in 2008 (104 mm) and site c was not surveyed between 2005 and 2007.

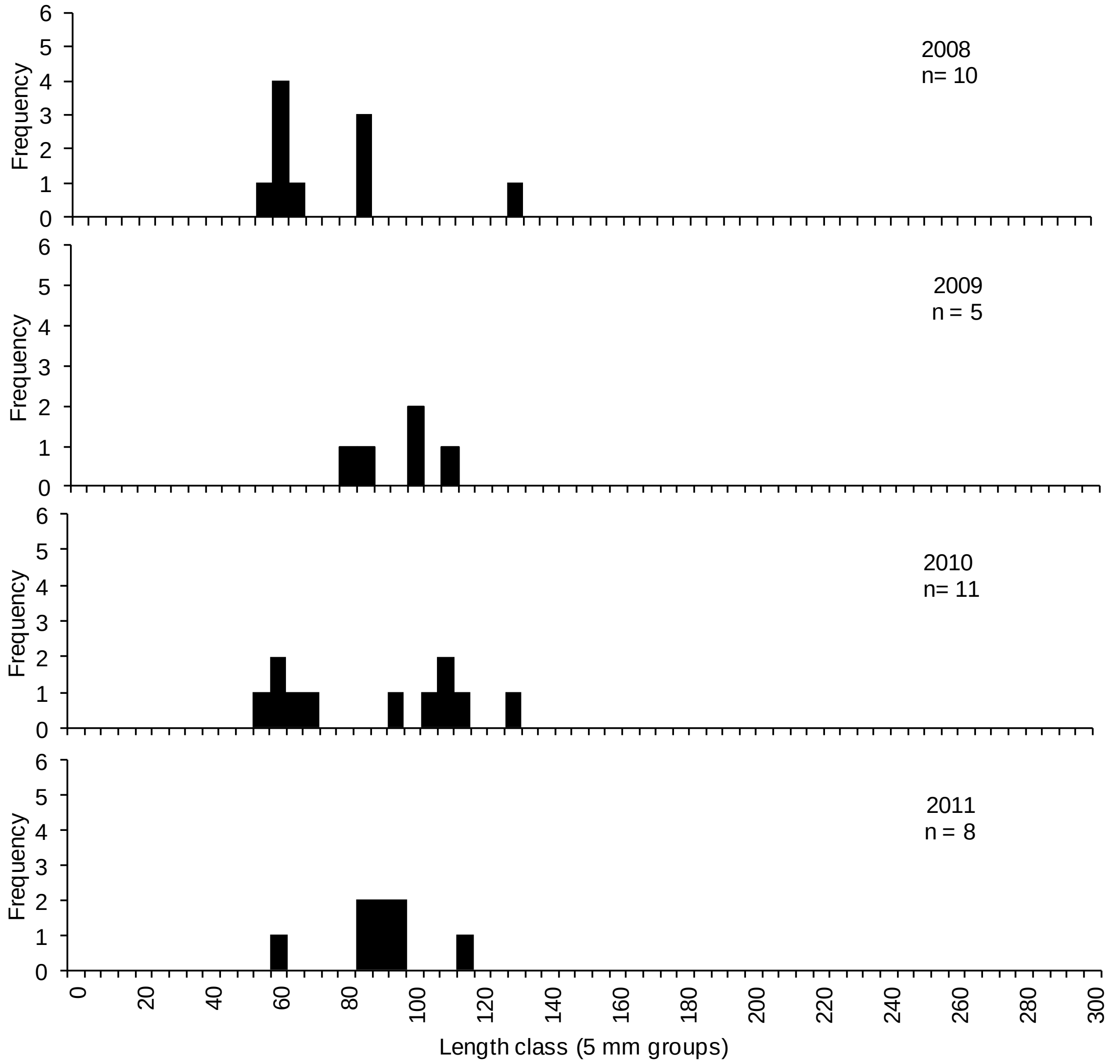

Figure 4.5 Length distributions of trout at site d in Kinnaird Burn between 2008 and 2011. Note site d was not surveyed between 2005 and 2007.

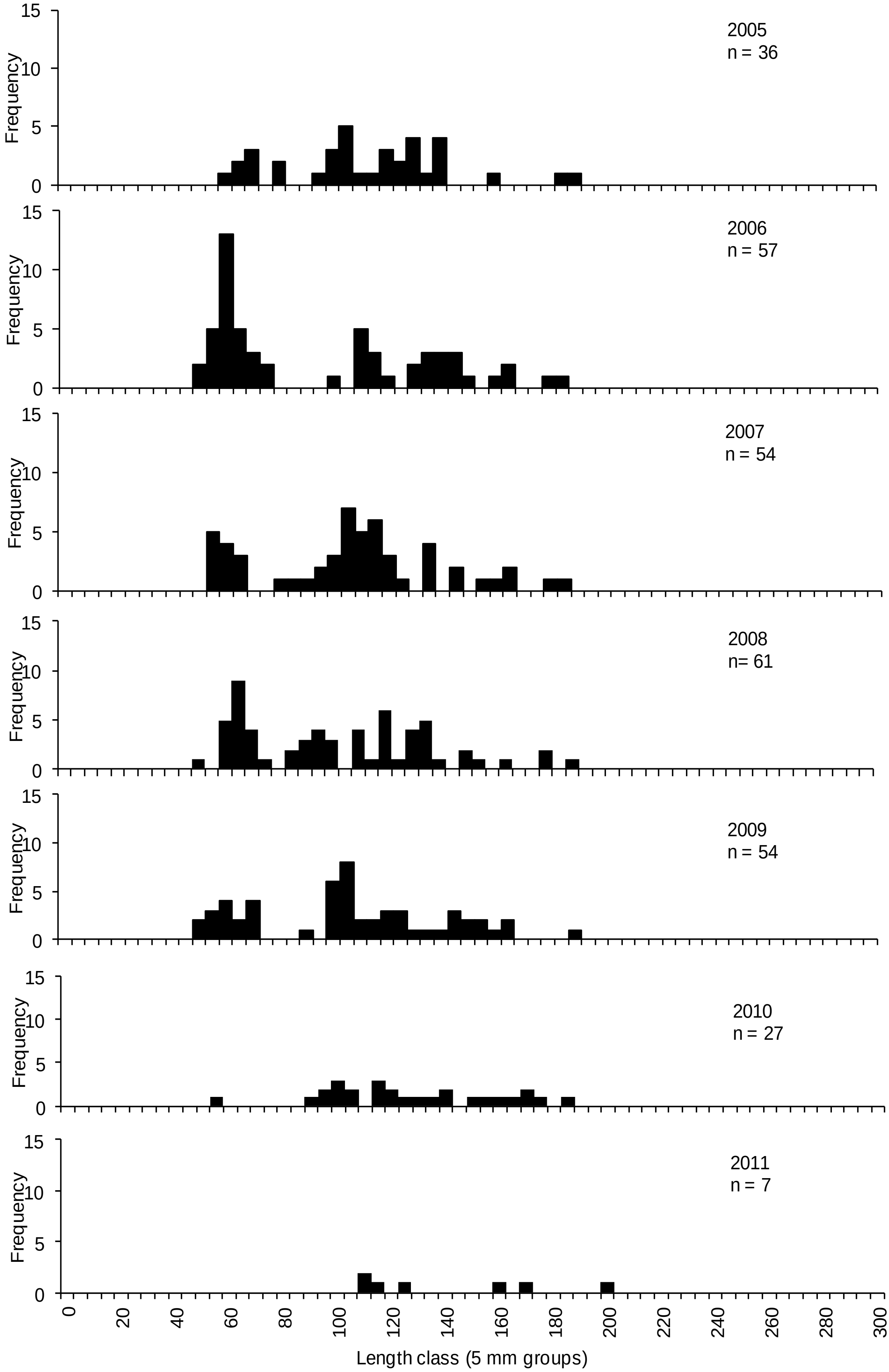

Figure 4.6 Length distributions of trout at site e in Kinnaird burn between 2005 and 2011.

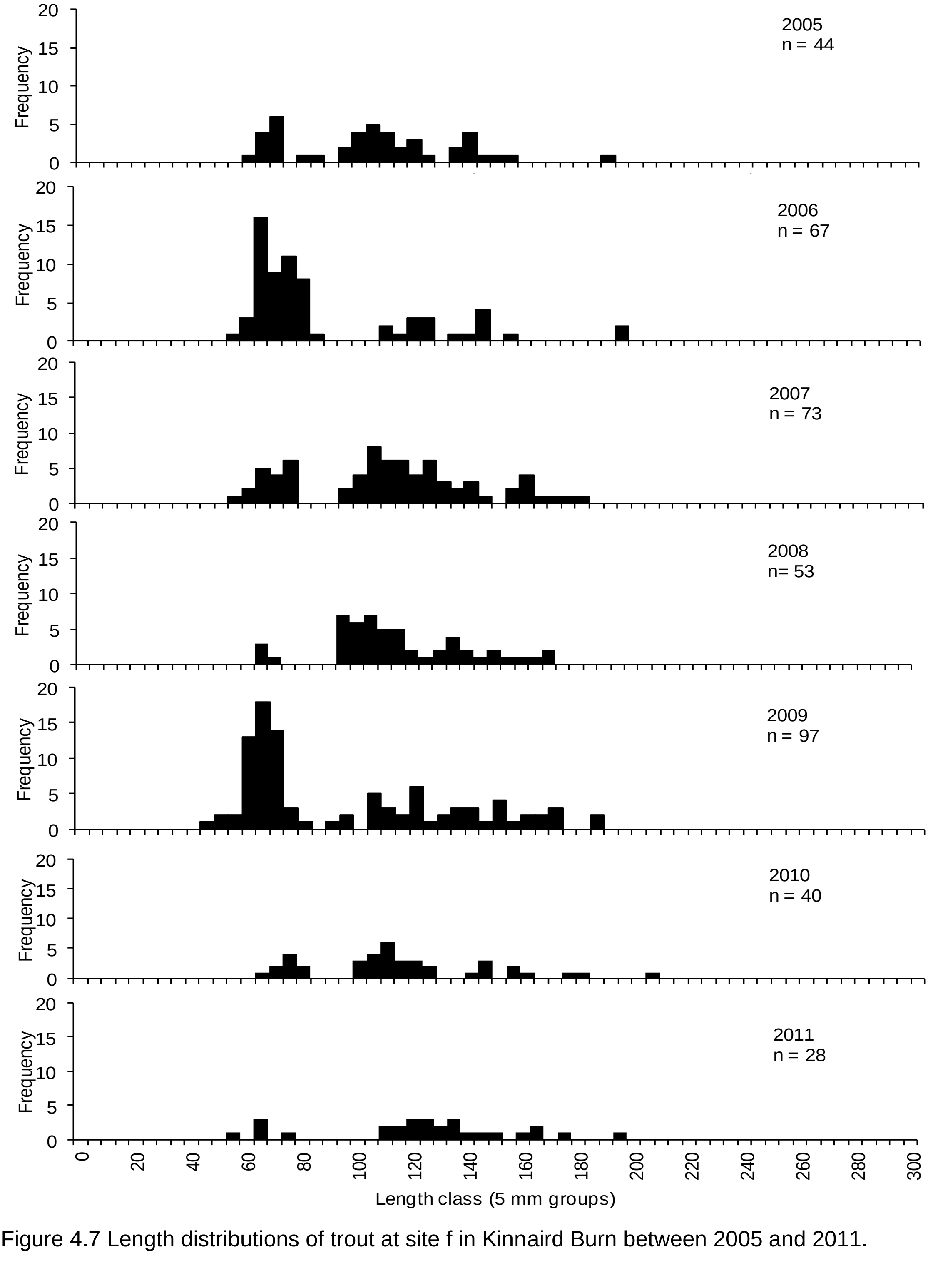

Figure 4.7 Length distributions of trout at site f in Kinnaird Burn between 2005 and 2011.

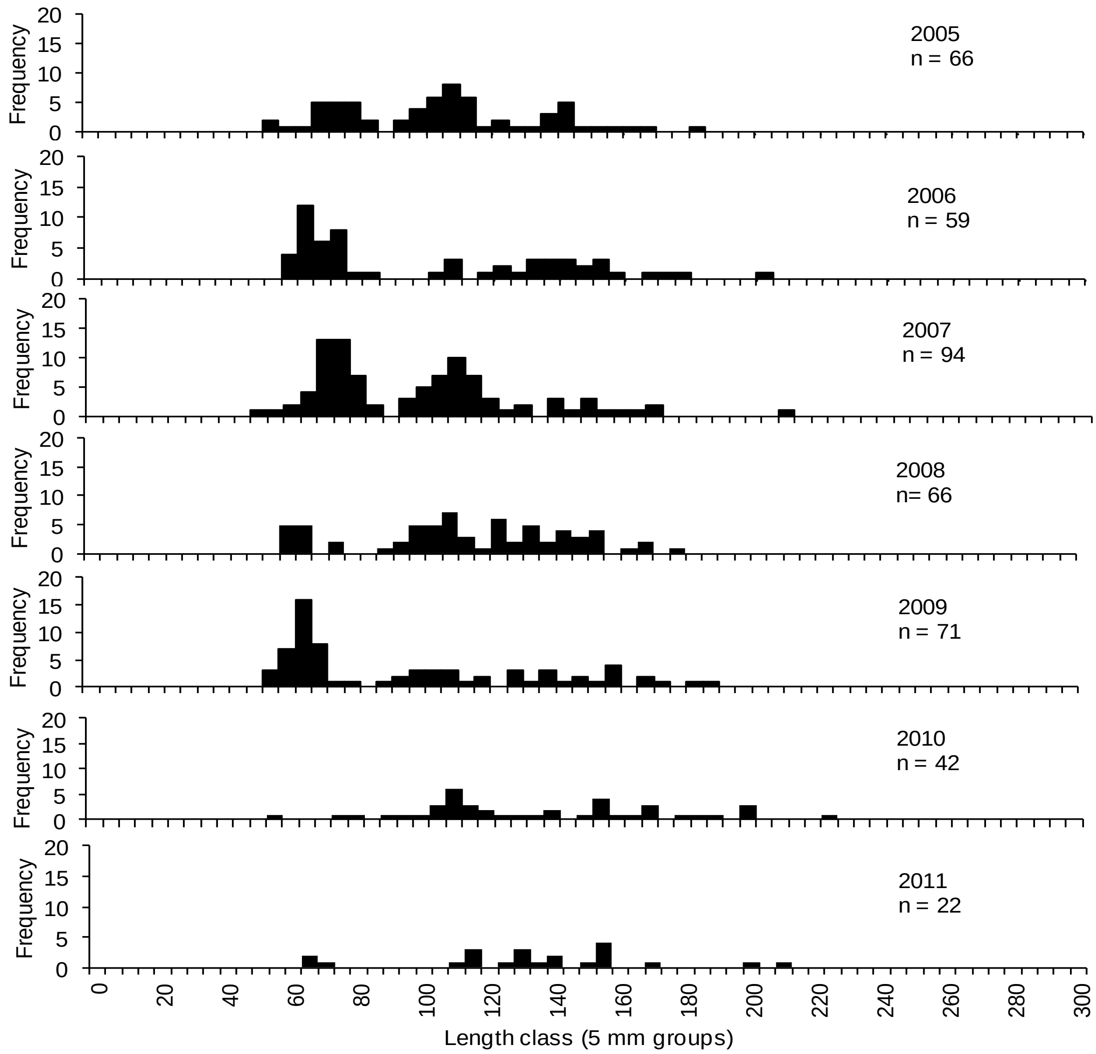

Figure 4.8 Length distributions of trout at site g in Kinnaird Burn between 2005 and 2011.

4.2.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was commissioned were compared to the same information collected following commissioning at two key sites in the lower reaches of Kinnaird Burn. For the analysis habitat data collected in 2008 were utilised with fisheries data collected from 2005-2008 (before commissioning) and compared to habitat data collected in 2011 utilised with fisheries data collected from 2009-2011 (after commissioning) (Tables 4.2 & 4.3). Although salmon were not captured in the surveys the species may be able to ascend to the lower reaches of the hydropower scheme should the damaged fish pass be reinstated, therefore HABSCORE analysis was performed for salmon to assess the potential of the reach. The data allowed comparison between one site in the abstracted reach and one site downstream of the abstracted reach.

The observed densities of 0+ salmon at site f (abstracted reach) both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were zero and hence lower than predicted by the Habitat Quality Score (HQS), suggesting poorer populations than expected; the Habitat Utilisation Index (HUI) upper CLs were <1 in both scenarios therefore the observed populations were significantly lower than expected (Table 4.2). The observed densities of ≥1+ salmon at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were also zero and hence lower than predicted by the HQS, suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.2).

The observed densities of 0+ trout at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.2). The observed density of ≥1+ trout (<20 cm) at site f prior to commissioning of the hydropower scheme (2005-2008 data) was higher than predicted (HQS), suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 12). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2009-2011 data) was higher than predicted (HQS), suggesting better populations than expected and the HUI lower CL was >1 therefore the observed population was significantly higher than expected (Table 4.2).

The observed densities of ≥1+ trout (>20 cm) at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.2).

The observed densities of 0+ salmon at site g both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were zero and hence lower than predicted by the HQS, suggesting poorer populations than expected; the HUI upper CLs were <1 in both scenarios therefore the observed populations were significantly lower than expected (Table 4.3). The observed densities of ≥1+ salmon at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were also zero and hence lower than predicted by the HQS, suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.3).

Table 4.2 HABSCORE outputs for site f (abstracted reach) before (2005-2008) and after (2009-2011) the Kinnaird Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2008	0.0	0.00	8.95	2.66	30.09	0.06	0.01	0.35	-2.81
2009-2011	0.0	0.00	28.25	7.43	107.45	0.02	0.00	0.16	-3.9
≥1+ salmon									
2005-2008	0.0	0.00	2.54	0.74	8.66	0.22	0.04	1.25	-1.51
2009-2011	0.0	0.00	1.97	0.57	6.83	0.30	0.05	1.90	-1.20
0+ trout									
2005-2008	15.7	8.60	11.80	3.05	45.70	0.73	0.11	4.94	-0.31
2009-2011	13.5	7.96	9.20	2.35	36.06	0.86	0.13	5.90	-0.15
≥1+ trout (<20 cm)									
2005-2008	35.7	19.53	3.32	0.77	14.27	5.88	0.95	35.38	1.77
2009-2011	32.2	18.96	2.16	0.49	9.57	8.80	1.42	54.39	2.17
≥1+ trout (>20 cm)									
2005-2008	0.0	0.00	1.10	0.35	3.47	0.50	0.16	1.58	-0.69
2009-2011	1.0	0.59	1.27	0.39	4.14	0.46	0.14	1.50	-0.78

Table 4.3 HABSCORE outputs for site g (control) before (2005-2008) and after (2009-2011) the Kinnaird Burn hydropower scheme was commissioned . (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2008	0.0	0.00	7.66	2.31	25.35	0.07	0.01	0.36	-2.65
2009-2011	0.0	0.00	4.59	1.35	15.61	0.10	0.01	0.77	-2.30
≥1+ salmon									
2005-2008	0.0	0.00	2.46	0.72	8.36	0.20	0.04	1.17	-1.60
2009-2011	0.0	0.00	1.74	0.51	5.99	0.27	0.04	1.74	-1.31
0+ trout									
2005-2008	24.8	12.41	10.94	2.84	42.20	1.13	0.17	7.66	0.12
2009-2011	7.0	3.35	7.32	1.88	28.46	0.46	0.07	3.12	
≥1+ trout (<20 cm)									
2005-2008	43.8	21.92	3.40	0.78	14.81	6.44	1.06	39.22	1.86
2009-2011	29.1	13.92	4.07	0.90	18.41	3.42	0.54	21.47	1.23
≥1+ trout (>20 cm)									
2005-2008	1.0	0.56	1.19	0.37	3.81	0.42	0.13	1.34	-0.86
2009-2011	1.3	0.60	2.09	0.62	6.99	0.29	0.09	0.92	-1.24

The observed density of 0+ trout prior to commissioning of the hydropower scheme (2005-2008 data) was higher than predicted (HQS) suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed populations were not significantly higher than expected (Table 4.3). The observed density of 0+ trout following commissioning of the hydropower scheme (2009-2011 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 13). The observed density of ≥1+ trout (<20 cm) prior to commissioning of the hydropower scheme (2005-2008 data) was higher than predicted (HQS) suggesting better populations than expected, and the HUI lower CL was >1 therefore the observed population was significantly higher than expected (Table 4.3). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2009-2011 data) was also higher than predicted (HQS) suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.3). The observed densities of ≥1+ trout (>20 cm) at site g both prior to and following commissioning of the hydropower scheme (2005-2008 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CL was <1 for the latter scenario indicating the observed population was significantly lower than expected (Table 4.3).

4.2.4 Salmonid growth rates

Upstream of the intakes

At site a, upstream of the main intake, back-calculated length of trout at age 1 ranged between 65 and 72 mm with observed length at age 1 in 2011 being slightly smaller at 60 mm (Table 4.4). Growth of trout in the first year of life (excluding 2011 data) was fastest in 2009 with the slowest first year growth in 2010, albeit the latter estimate is based on one fish (Table 4.4); back-calculated length of trout at age 2 ranged between 106 and 114 mm (Table 4.4).

At site c, upstream of the intake on the small west tributary, back-calculated length of trout at age 1 ranged between 59 and 67 mm with observed length at age 1 in 2011 being 52 mm (Table 4.4). Growth of trout in the first year of life (excluding 2011 data) was slowest in 2007 and fastest in 2009 (Table 4.4).

Depleted reach

At site b, downstream of the main intake, back-calculated length of trout at age 1 ranged between 51 and 75 mm with observed length at age 1 in 2011 being 58 mm (Table 4.4). Growth of trout in the first year of life was slowest in 2001 and 2002 with the fastest first year growth in 2010 (Table 4.4). Back-calculated length of trout at age 2 ranged between 90 and 137 mm, with the slowest growth in 2001 and fastest in 2005 (Table 4.4). Back-calculated length of trout aged 3 ranged between 124 and 158 mm, with the slowest growth in 2001 and fastest in 2003 (Table 4.4). Back-calculated length of trout aged 4 ranged between 152 and 180 mm, and at age 5 was 207 mm in 2000 (Table 4.4). It should be noted that the last few year groups are based on small numbers of fish, often only one individual, so are subject to wide variation.

At site d, downstream of the intake on the small west tributary, back-calculated length of trout at age 1 ranged between 56 and 71 mm with observed length at age 1 in 2011 being 57 mm (Table 4.4). Growth of trout in the first year of life was slowest in 2010 with the fastest first year growth occurring in 2006 (Table 4.4). Back-calculated length of trout at age 2 ranged between 107 and 120 mm, with the slowest growth in 2005 and fastest in 2006 (Table 4.4).

At site e, in the middle reaches of the abstracted reach, back-calculated length of trout at age 1 ranged between 50 and 72 mm (Table 4.4). Growth of trout in the first year of life was slowest in 2002 with the fastest first year growth occurring in 2008 and 2009 (Table 4.4). Back-calculated length of trout at age 2 ranged between 105 and 127 mm, with the slowest growth in 2002 and fastest in 2009 (Table 4.4). Back-calculated length of trout aged 3 ranged between 132 and 155mm, with the slowest growth in 2005 and fastest in 2002 (Table 4.4).

At site f, upstream of the outfall, back-calculated length of trout at age 1 ranged between 51 and 75 mm with observed length at age 1 in 2011 being 63 mm (Table 4.4). Growth of trout in the first year of life was slowest in 2003 and fastest in 2010 (Table 4.4). Back-calculated length of trout at age 2 ranged between 106 and 131 mm, with the slowest growth rate in 2003 and fastest in 2009 (Table 4.4). Back-calculated length of trout at age 3 ranged between 148 and 163 mm, with the slowest growth in 2002 and fastest in 2001 (Table 4.4).

Downstream of the outfall

At site g, downstream of the outfall, back-calculated length of trout at age 1 ranged between 59 and 75 mm with observed length at age 1 in 2011 being 65 mm (Table 4.4). Growth of trout in the first year of life was slowest in 2003 with the fastest first year growth occurring in 2009 (Table 4.4). Back-calculated length of trout at age 2 ranged between 105 and 116 mm, with the slowest growth in 2006 and fastest in 2009 (Table 4.4).

Table 4.4 Back-calculated lengths at age of trout in the upper and lower reaches of Kinnaird Burn. Note no 0+ trout were captured in 2011 at site e. Sites f and g are excluded from 2007 ageing data set as sites were combined and therefore not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))				
		1	2	3	4	5
a	2007	67±9(6)	106(1)			
	2008	68±5(9)	114±0(2)			
	2009	65±7(8)				
	2010	72(1)				
	*2011	60±7(3)				
b	2000	63(1)	108(1)	144(1)	180(1)	207(1)
	2001	51±3(4)	90±4(4)	124±5(4)	152±7(4)	
	2002	51±10(5)	102±8(5)	134±8(5)		
	2003	59±7(9)	106±11(9)	158(1)		
	2004	66±9(11)	116±15(6)			
	2005	72±16(15)	137(1)			
	2006	63±7(17)	117±11(2)			
	2007	68±11(16)	110±0(2)			
	2008	75±8(7)				
	2009	69±9(19)	109(1)			
	2010	76±9(5)				
	*2011	58±7(3)				
c	2007	59±12(6)	125(1)			
	2008	65±4(3)				
	2009	67±9(4)				
	2010	66±19(6)				
	*2011	52±6(2)				
d	2005	64(1)	107(1)	150(1)		
	2006	71±9(3)	120±7(3)			
	2007	66±10(17)				
	2008	61±10(4)				
	2010	56±8(7)				
	*2011	57(1)				
e	2001	62(1)	114(1)	155(1)	186(1)	
	2002	50±3(2)	105±0(2)	138±1(2)		
	2003	56±8(13)	112±13(13)			
	2004	65±8(19)	110±11(13)			
	2005	61±9(13)	105±14(7)	132±2(2)		
	2006	61±8(19)	108±8(6)	137(1)		
	2007	64±8(15)	118±5(5)			
	2008	72±11(14)	114±1(2)			
	2009	72±12(19)	127±1(2)			
	2010	67±7(4)				
f	2001	70(1)	116(1)	163(1)	186(1)	
	2002	56±5(3)	115±6(3)	148 ±7(3)		
	2003	51±9(9)	106±9(3)			
	2004	69±7(10)	111±7(3)			
	2005	67±12(11)				
	2006	59(1)	104(1)			
	2007	61±7(8)	112±10(7)	165(1)		
	2008	68±9(16)	116±17(4)			
	2009	72±9(19)	131±7(5)			
	2010	75±10(11)				
	*2011	63±6(5)				
g	2002	62±7(5)	107±8(5)	150±15(5)		
	2003	59±10(11)	107±15(11)			
	2004	64±10(20)	111±13(10)			
	2005	66±7(7)				

Table 4.4 (continued).

Site code	Year class	Back-calculated length at age (mm ± S.D (n))				
		1	2	3	4	5
g	2006	62(1)	105(1)	155(1)		
	2008	73±10(8)	114±17(2)			
	2009	75±30(21)	116±11(5)			
	2010	69±5(9)				
	*2011	65±1(3)				

It should be noted that the last few year groups at the sites surveyed were often based on small numbers of fish, often only one individual, so were subject to wide variation.

Overall during the study period the growth of trout within the abstracted reaches followed similar trends to the growth of trout at sites outside the abstracted reach.

4.2.5 Eel

Eels were absent from the upper and middle reaches of Kinnaird Burn throughout the study period. Only the occasional eel was captured at sites f and g suggesting the lower reaches of Kinnaird Burn are only occasionally used.

4.2.6 Wetted widths

A paired two sample *t*-test was performed on the mean wetted width (collected during HABSCORE) at site f, as this was located within the abstracted reach. Essentially, the test was used to distinguish if there was any difference between wetted widths each time HABSCORE was performed to ensure comparison of density estimates over the study period. The 95% confidence limits (-0.425, 0.775) enclosed zero with a P value of 0.557, n = 40; therefore there was no significant difference between wetted widths collected during the surveys pre and post abstraction, confirming comparability of density estimates.

4.3 Keltney Burn hydropower scheme

4.3.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

It should be noted that site a was not surveyed prior to the commissioning of the hydropower scheme due to relocation of the intake upstream of the original monitoring sites. However site a acts as a control site in 2010 and 2011 to allow comparison with sites b and c located in the abstracted reach; sites b and c were monitored prior to commissioning of the hydropower scheme.

Overall ≥1+ trout dominated populations in the upper reaches of Keltney Burn; salmon were absent from the upper reaches throughout the study period due to an impassable natural barrier in Keltney Burn village (Tables 4.6 & 4.7).

Upstream of the intake

0+ trout densities at site a, were poor (class E) in 2010 and 2011 (Figure 4.9, Table 4.5); ≥1+ trout density at site a was poor (class E) in 2011 and ≥1+ trout were absent in 2010.

Depleted reach

0+ and ≥1+ trout densities were also low or zero at site b, downstream of the intake, throughout the study period (Figure 4.9). 0+ trout were only captured in 2006 and 2010 and populations were classified as poor (class E). ≥1+ trout populations were highest in 2007 (Figure 4.9) and classified as fair/average (class C) while in 2005 and 2010 populations were poor (class E); ≥1+ trout were absent in 2006 and 2011 (Table 4.5).

At site c, downstream of the intake, 0+ and ≥1+ trout densities were low or zero throughout the study period; 0+ trout were only captured in 2005 and populations were poor (class E) (Figure 4.9, Table 4.5). ≥1+ trout populations were highest in 2007, as found at site b, and classified as fair/average (class C) while in other years populations were poor (class E) (Figure 4.9, Table 4.5).

Catches of salmonids in the lower reaches of Keltney Burn were dominated by salmon with lower densities of trout (Table 4.5).

At site d, upstream of the outfall, 0+ and ≥1+ trout populations remained stable throughout the study period and were classified as poor (class E) (Figure 4.9, Table 4.5). 0+ salmon densities between 2005 and 2008 varied from good (class B) to fair/average (class C); between 2009 and 2011 populations were stable and fair/average (class C) (Figure 4.9, Table 4.6). ≥1+ salmon populations were generally stable throughout the study period varying between excellent (class A) and good (class B) (Figure 4.9, Table 4.6). Assessment of densities of ≥1+ salmon using the other SFCC classification schemes revealed variations, with populations similar or one grade lower in other years using the regional/river width specific classification schemes (Table 4.6); the EA-FCS classified ≥1+ salmon populations as excellent (class A) throughout the study period.

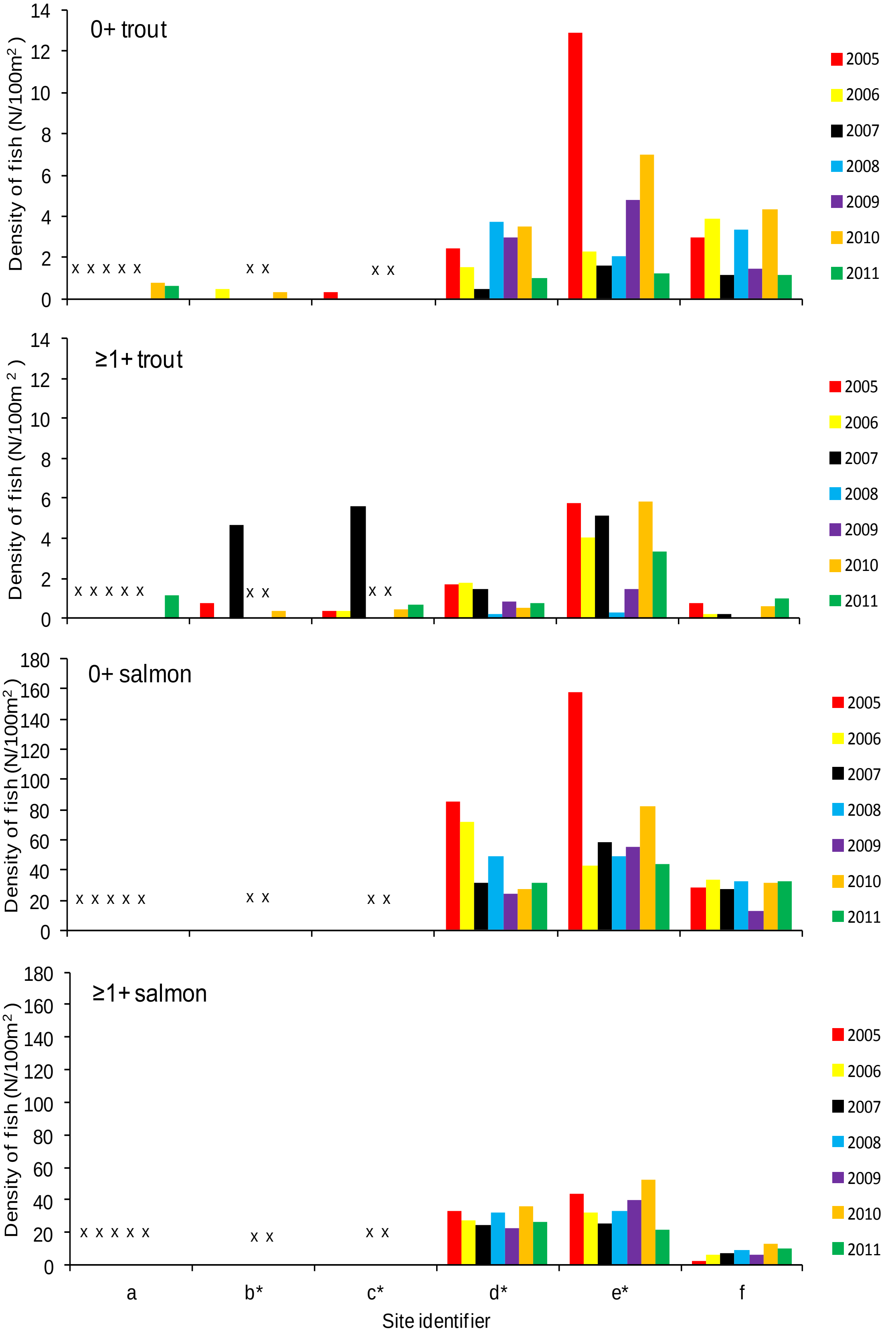

Figure 4.9 Density estimates of trout and salmon in Keltney Burn between 2005 and 2011 (Note, site a was not sampled from 2005-2009 and sites b-c were not sampled in 2008 and 2009). Hydropower scheme was commissioned one month before 2010 fish surveys. x represents years not surveyed.

Table 4.5 Classification of trout densities in both the upper and lower reaches of Keltney Burn between 2005 and 2011 based on five classification schemes. Note hydropower scheme commissioned one month before 2010 fish surveys. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2005	n/s	n/s	n/s	n/s	n/s
	2006	n/s	n/s	n/s	n/s	n/s
	2007	n/s	n/s	n/s	n/s	n/s
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	E	E	E	E	E
b*	2005	F	F	F	F	F
	2006	E	E	E	E	E
	2007	F	F	F	F	F
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	F	F	F	F	F
c*	2005	E	E	E	E	E
	2006	F	F	F	F	F
	2007	F	F	F	F	F
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	F	F	F	F	F
	2011	F	F	F	F	F
d*	2005	E	E	E	E	E
	2006	E	E	E	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	D
	2009	E	E	C	D	E
	2010	E	E	C	D	D
	2011	E	E	E	E	E
e*	2005	D	E	C	D	C
	2006	E	E	E	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2009	E	E	E	E	D
	2010	D	E	D	E	D
	2011	E	E	E	E	E
f	2005	E	E	C	D	D
	2006	E	E	E	E	D
	2007	E	E	E	E	E
	2008	E	E	D	E	D
	2009	E	E	E	E	E
	2010	D	E	C	D	D
	2011	E	E	E	E	E

≥1+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2005	n/s	n/s	n/s	n/s	n/s
	2006	n/s	n/s	n/s	n/s	n/s
	2007	n/s	n/s	n/s	n/s	n/s
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	F	F	F	F	F
	2011	E	E	C	C	E
b*	2005	E	E	E	E	E
	2006	F	F	F	F	F
	2007	C	C	C	D	D
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	F	F	F	F	F
c*	2005	E	E	E	E	E
	2006	E	E	E	E	E
	2007	C	C	C	D	C
	2008	n/s	n/s	n/s	n/s	n/s
	2009	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	E	E	E	E	E
d*	2005	E	E	D	E	E
	2006	E	E	D	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2009	E	E	E	E	E
	2010	E	E	E	E	E
	2011	E	E	E	E	E
e*	2005	C	C	C	C	C
	2006	D	E	D	E	D
	2007	C	C	B	B	C
	2008	F	F	F	F	E
	2009	E	E	E	E	E
	2010	C	C	B	B	C
	2011	D	D	C	D	E
f	2005	E	E	D	C	E
	2006	F	F	F	F	E
	2007	E	E	E	E	E
	2008	F	F	F	F	F
	2009	F	F	F	F	F
	2010	E	E	D	E	E
	2011	E	E	C	C	E

At site e, upstream of the outfall, 0+ trout populations were classified as poor (class E) between 2006 and 2009 and in 2011 but were fair/poor class (D) in other years (Figure 4.9, Table 4.5). ≥1+ trout populations varied considerably over the study period ranging from fair/average (class C) in 2005, 2007 and 2010 to absent (class F) in 2008 (Figure 4.9, Table 4.5). 0+ salmon densities were greatest in 2005 but generally stable in other survey years (Figure 4.9); 0+ salmon populations were excellent (class A) in 2005 and 2010, good (class B) between 2007 and 2009 and fair/average (class C) in other years (Table 4.6). Assessment of densities of 0+ salmon using the other SFCC classification schemes revealed variations, with the populations similar or one grade lower in other years using the regional/river width specific classification schemes (Table 4.6); the EA-

FCS classifications were identical to the SFCC national scheme except in 2010. ≥1+ salmon populations were generally stable throughout the study period and predominantly excellent (class A), except in 2006 when populations were fair/average (class C) and in 2011 when populations were good (class B) (Figure 4.9, Table 4.6).

Table 4.6 Classification of salmon densities in the lower reaches of Keltney Burn between 2005 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
d*	2005	B	C	B	C	B
	2006	C	D	C	D	B
	2007	C	D	C	D	C
	2008	B	C	B	D	B
	2009	C	D	C	D	C
	2010	C	D	C	D	C
	2011	C	D	C	D	C
e*	2005	A	B	A	B	A
	2006	C	D	C	D	C
	2007	B	C	B	C	B
	2008	B	C	B	C	B
	2009	B	C	B	C	B
	2010	A	B	A	C	B
	2011	C	D	C	D	C
f	2005	C	D	C	D	C
	2006	C	D	C	D	C
	2007	C	D	C	D	C
	2008	D	D	D	D	C
	2009	D	E	D	D	D
	2010	C	D	C	D	C
	2011	C	D	C	D	C

≥1+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
d*	2005	A	A	A	B	A
	2006	B	B	B	B	A
	2007	A	B	A	B	A
	2008	A	B	A	B	A
	2009	B	C	B	C	A
	2010	A	A	A	A	A
	2011	B	B	B	B	A
e*	2005	A	A	A	A	A
	2006	C	C	C	D	A
	2007	A	B	A	B	A
	2008	A	B	A	B	A
	2009	A	A	A	B	A
	2010	A	A	A	A	A
	2011	B	B	B	C	A
f	2005	E	E	E	E	E
	2006	D	D	C	D	C
	2007	D	D	D	D	C
	2008	D	D	D	D	C
	2009	D	D	C	D	C
	2010	C	C	B	C	B
	2011	D	D	C	D	B

Downstream of the outfall

At site f, located adjacent to and downstream of the outfall, densities of salmon were overall lower compared to those observed at sites d and e (Figure 4.9), reflecting the generally different flow/habitat characteristics at site f. 0+ salmon populations varied between fair/average (class C) and fair/poor (class D) throughout the study period (Table 4.6). ≥1+ salmon populations were predominantly fair/poor (class D) except in 2005 when populations were poor (class E) and 2010 when populations were fair/average (class C). Trout populations were dominated by 0+ individuals; 0+ trout populations were poor (class E) in all years except 2010 (Figure 4.9, Table 4.5). ≥1+ trout populations were poor (class E) or absent (class F) during the study period (Figure 4.9, Table 4.5).

4.3.2 Salmonid size structure

Upstream of the intake

The low numbers of trout captured at sites in the upper reaches throughout the study period make identification of trends in size structure difficult however the following describes any general trends observed. Length distributions of brown trout in the upper reaches associated with the intake varied between years at each site.

At site a, upstream of the main intake, 0+ trout were caught in small numbers in the size range 55-70 mm while ≥1+ trout in 2011 were caught in the size range 100-160 mm.

Depleted reach

At site b, downstream of the intake, 0+ trout were caught in the size range 70-80 mm during the study period; ≥1+ trout were caught in the size range 100-180 mm. At site c, only one 0+ trout in the size range 55-60 mm was caught; ≥1+ trout were caught in the size range 75-170 mm during the study period.

Length distributions of salmon in the lower reaches of Keltney Burn were similar between years at each site (Figures 4.10-4.12). At site d, upstream of the outfall 0+ salmon were caught in the size range 35-75 mm while ≥1+ salmon were caught in the size range 80-125 mm, throughout the study period except in 2006 where the largest salmon caught was 160 mm (Figure 4.10). At site e, upstream of the outfall, 0+ salmon were caught in the size range 35-75 mm throughout the study period while ≥1+ salmon were caught in the size range 80-130 mm except in 2008 when the largest salmon caught was 146 mm (Figure 4.11).

At site d, 0+ trout were caught in the size range 45-80 mm throughout the study period, albeit in small numbers in 2007 and 2011 (Figure 4.13). ≥1+ trout were caught in a similar size range (80-170 mm) throughout the study period except in 2006 when one large individual >180 mm was captured (Figure 4.13). At site e, 0+ trout were caught in the size range 45-80 mm throughout the study period, albeit in low numbers in some years (Figure 4.14). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2007, 2010 and 2011 when small numbers of larger individuals >180 mm were captured (Figure 4.14).

Downstream of the outfall

At site f, located adjacent to and downstream of the outfall, 0+ salmon were caught in the size range 40-70 mm throughout the study period while ≥1+ salmon were caught in the size range 80-130 mm (Figure 4.12). At site f, located adjacent to and downstream of the outfall, 0+ trout were caught in the size range 55-80 mm throughout the study period, albeit in low numbers in some years (Figure 4.15); ≥1+ trout were caught in small numbers and in a narrow size range of 110-155 mm throughout the study (Figure 4.15).

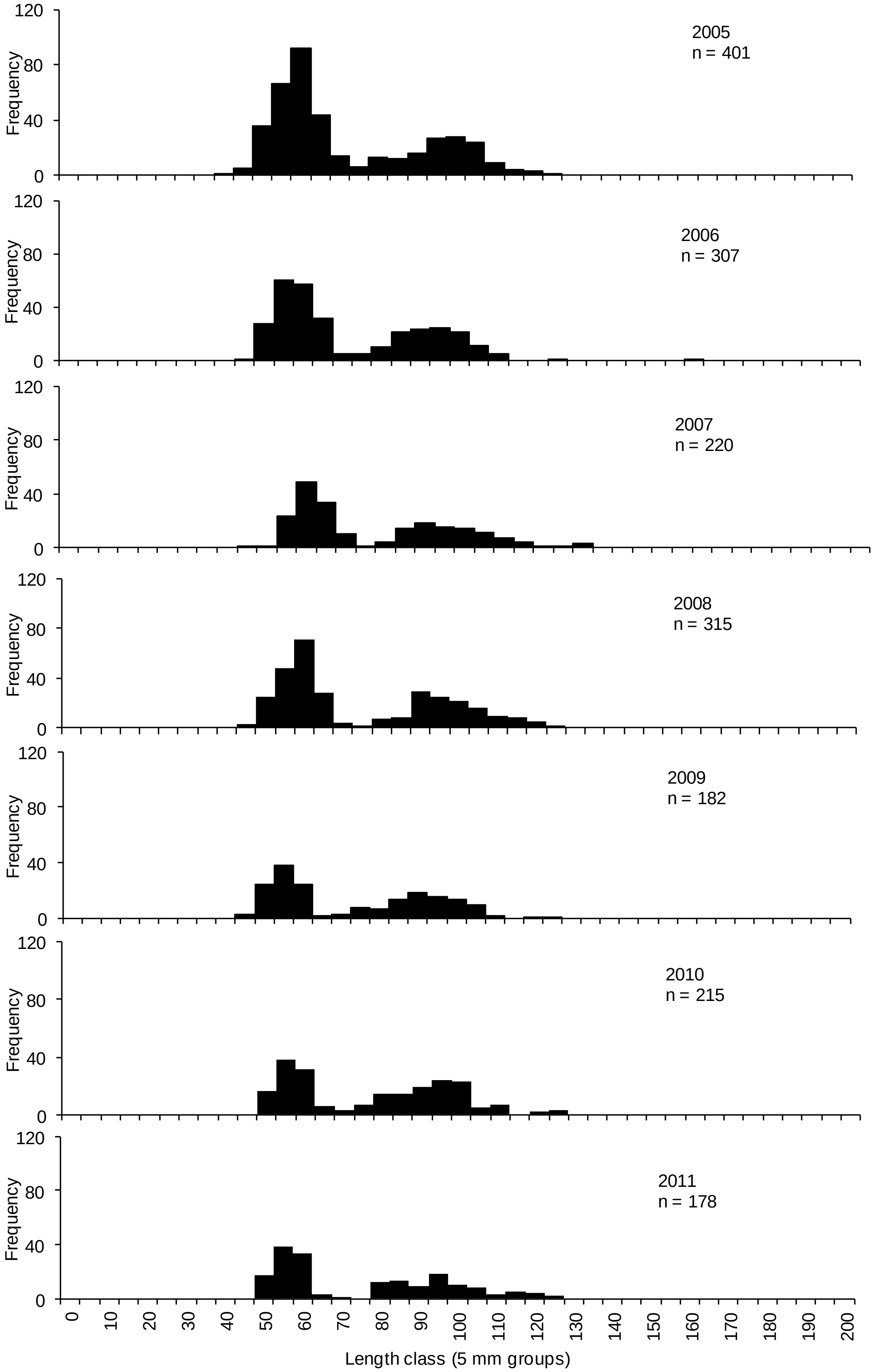

Figure 4.10 Length distributions of salmon at site d in Keltney Burn between 2005 and 2011.

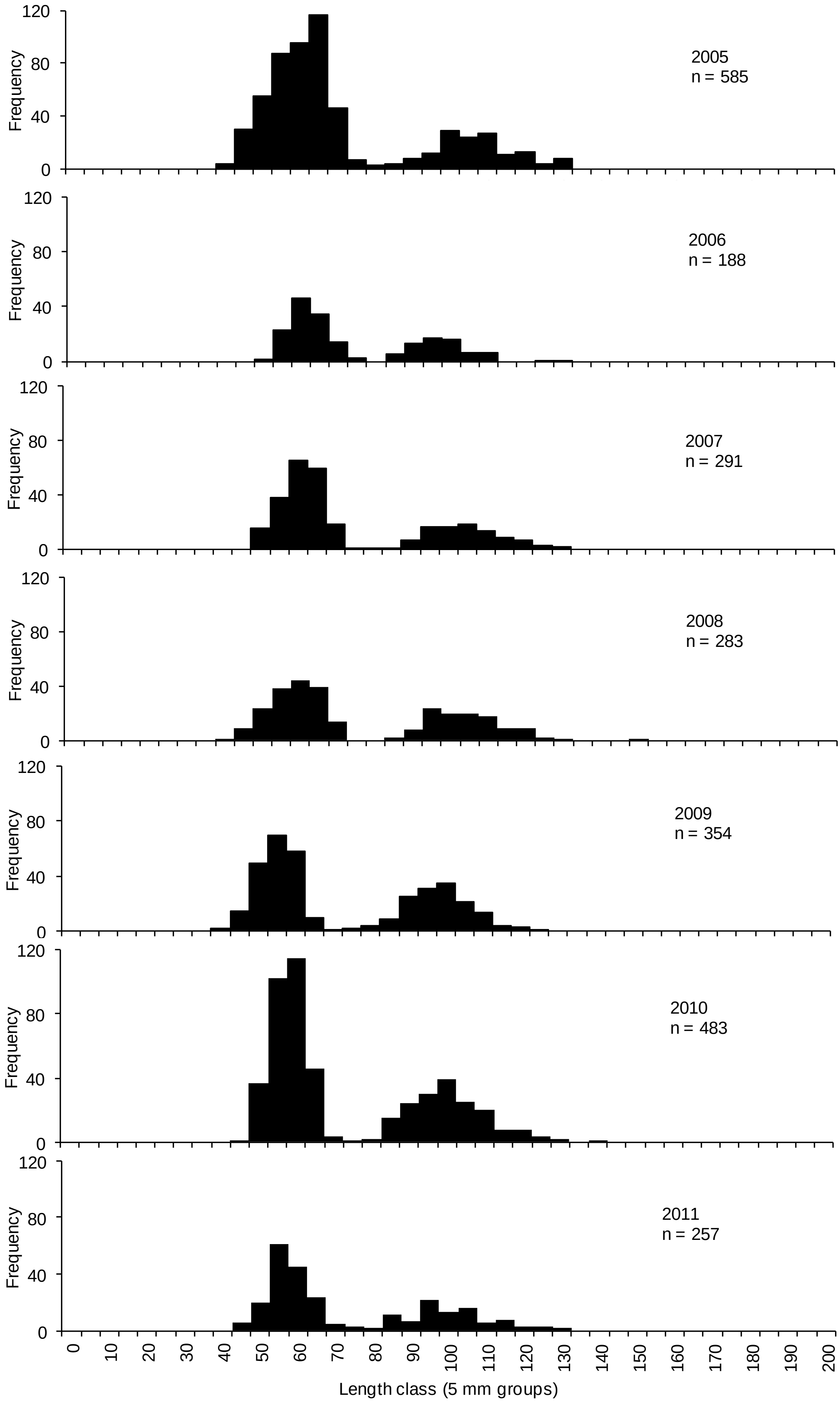

Figure 4.11 Length distributions of salmon at site e in Keltney Burn between 2005 and 2011.

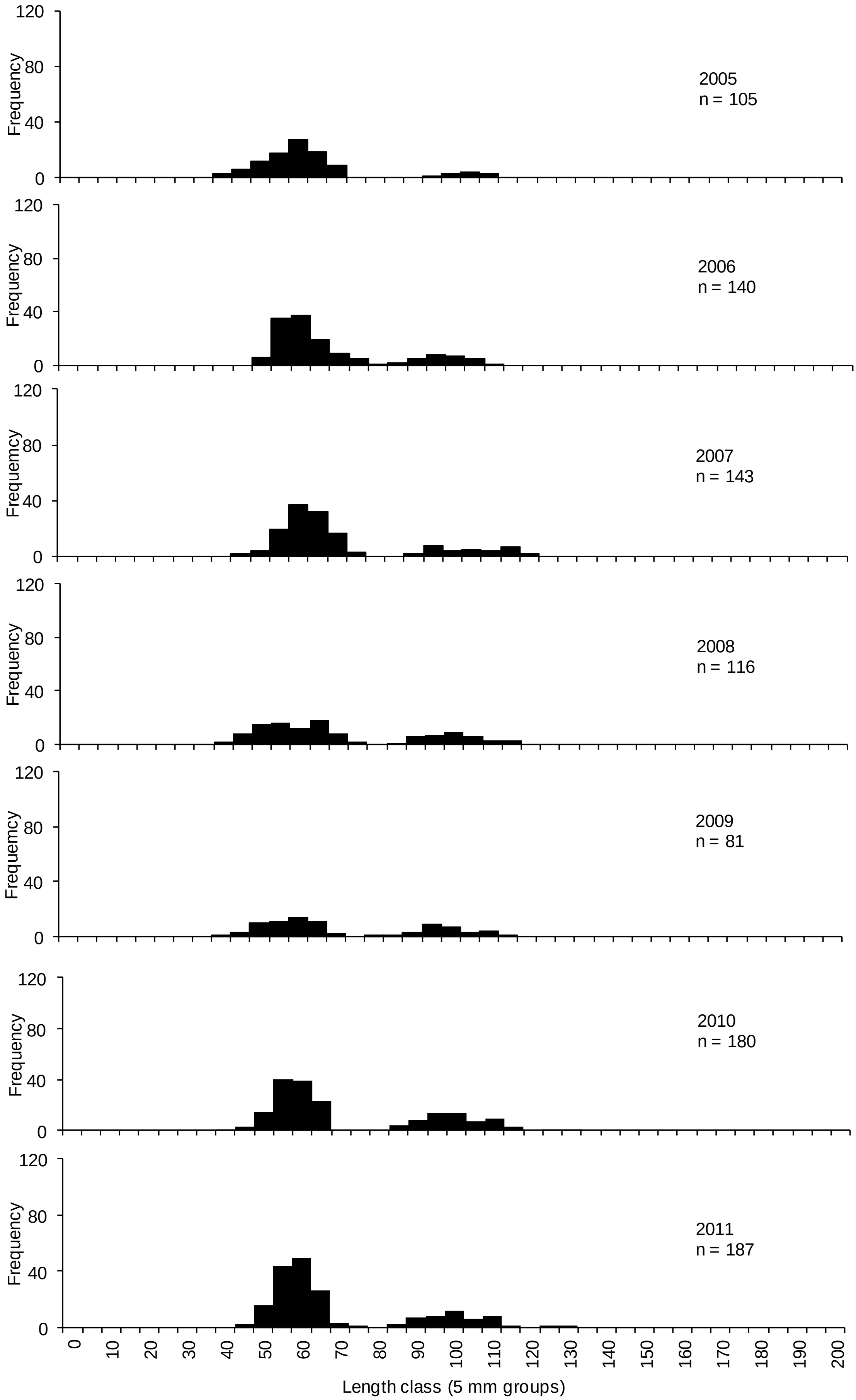

Figure 4.12 Length distributions of salmon at site f in Keltney Burn between 2005 and 2011.

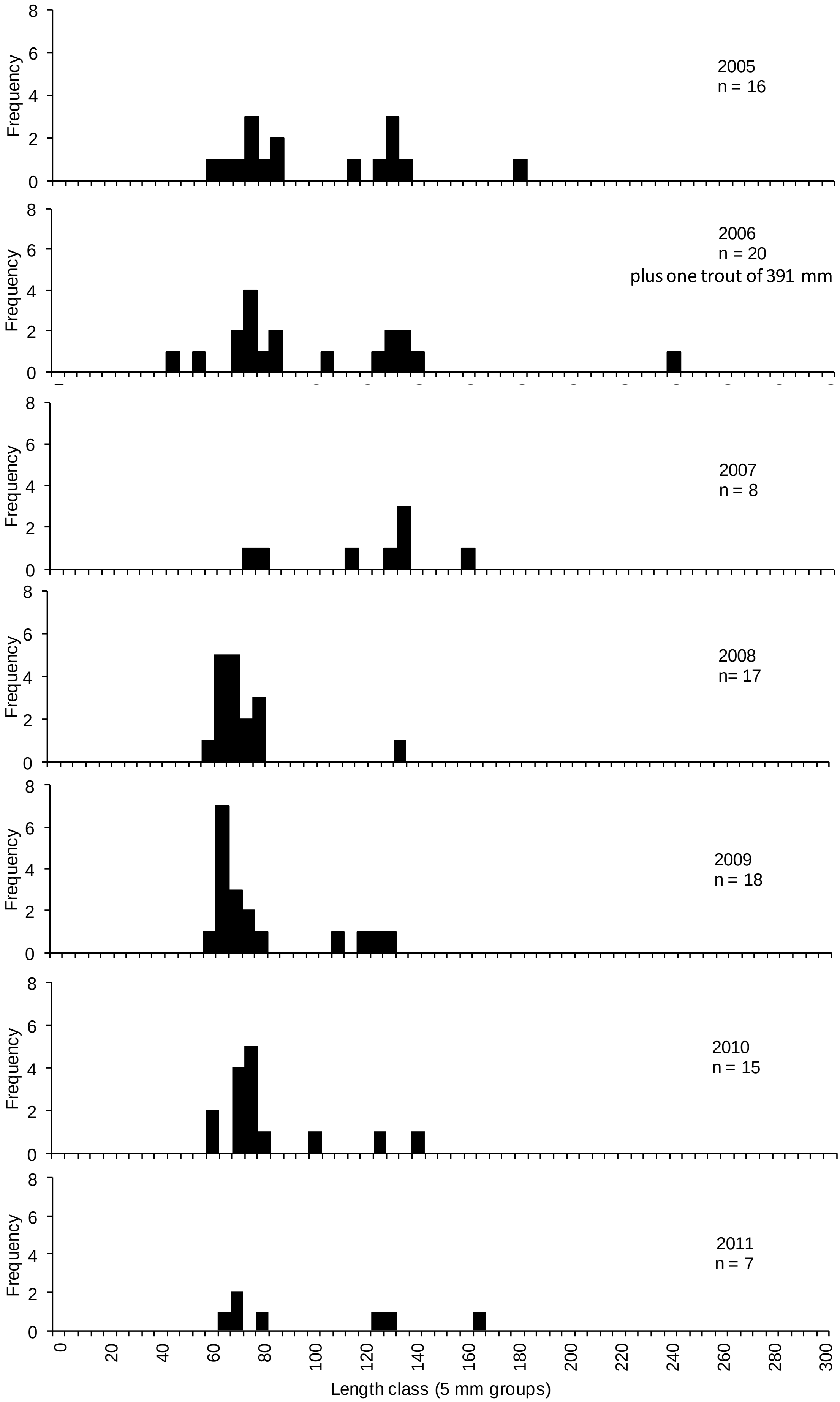

Figure 4.13 Length distributions of trout at site d in Keltney Burn between 2005 and 2011.

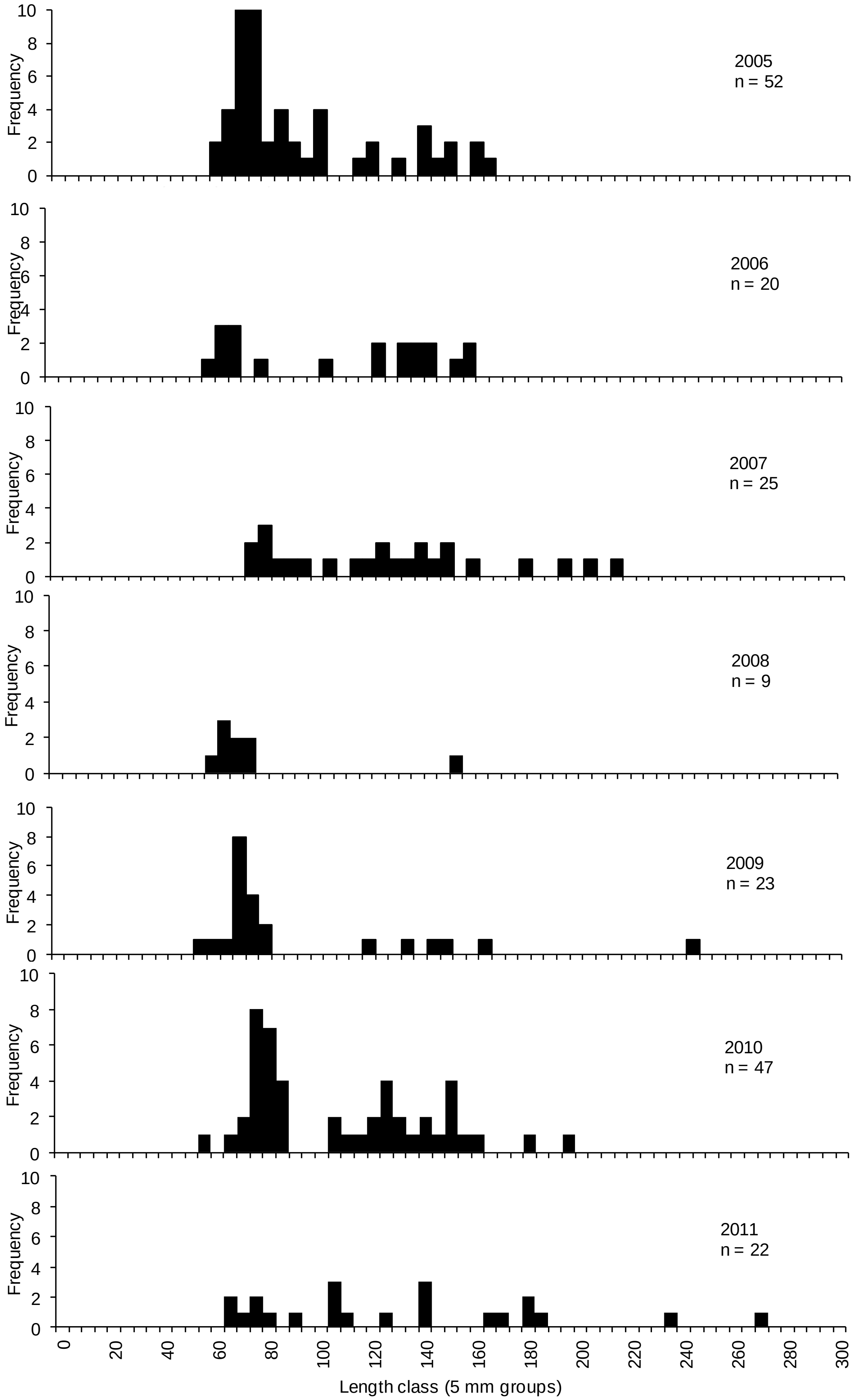

Figure 4.14 Length distributions of trout at site e in Keltney Burn between 2005 and 2011.

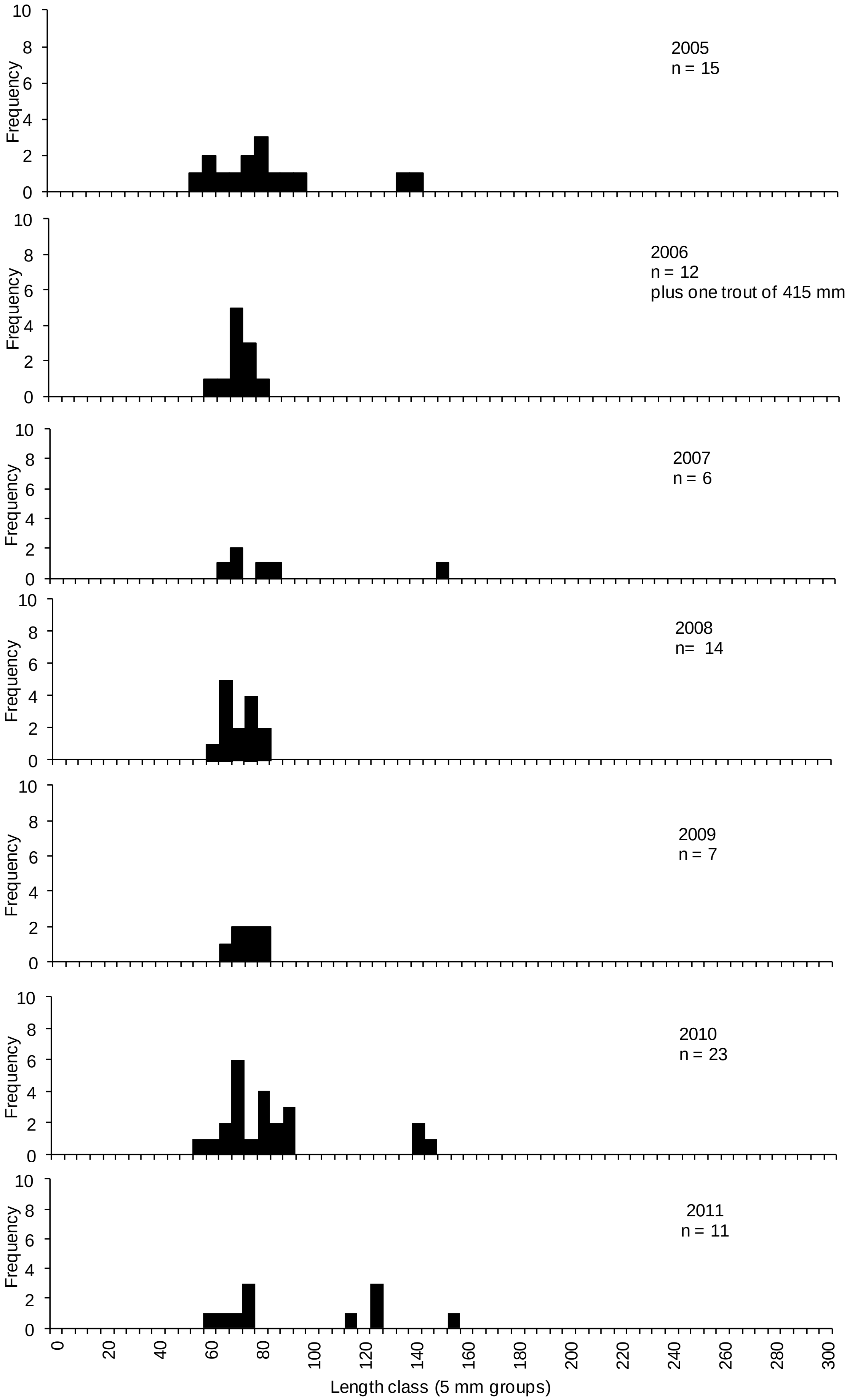

Figure 4.15 Length distributions of trout at site f in Keltney Burn between 2005 and 2011.

4.3.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was commissioned were compared to the same information collected following commissioning at three key sites in the lower reaches of Keltney Burn. For the analysis habitat data collected in 2009 were utilised with fisheries data collected from 2005-2009 and compared to habitat data collected in 2011 utilised with fisheries data collected from 2010-2011 (Tables 4.7-4.9). The data allowed comparison between sites within the abstracted reach (sites d and e) and one site (site f) adjacent to and downstream of the abstracted reach.

The observed density of 0+ salmon at site d prior to commissioning of the hydropower scheme (2005-2009 data) was marginally higher than predicted by the Habitat Quality Score (HQS), suggesting better populations than expected but the HUI lower CL was <1, therefore the observed population was not significantly higher than expected (Table 4.7). The observed density of 0+ salmon following commissioning of the hydropower scheme (2010-2011 data) was marginally lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.7). The observed densities of ≥1+ salmon at site d both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected and the HUI lower CLs were >1 in both scenarios therefore the observed populations were significantly higher than expected (Table 4.7). The observed densities of 0+ trout at site d both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.7). The observed density of ≥1+ trout (<20 cm) at site d prior to commissioning of the hydropower scheme (2005-2009 data) was marginally higher than predicted (HQS), suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.7). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2010-2011 data) was marginally lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than would be expected (Table 4.7). The observed densities of ≥1+ trout (>20 cm) at site d both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than

Table 4.7 HABSCORE outputs for site d (abstracted reach) before (2005-2009) and after (2010-2011) the Keltney Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2009	206.8	43.68	40.45	10.00	163.60	1.08	0.13	8.91	0.07
2010-2011	109.2	29.15	33.00	8.45	128.86	0.88	0.11	7.13	-0.13
≥1+ salmon									
2005-2009	119.1	25.15	1.45	0.40	5.27	17.29	2.64	113.42	2.85
2010-2011	115.3	30.79	1.91	0.53	6.83	16.13	2.48	105.06	2.78
0+ trout									
2005-2009	7.9	1.66	3.97	6.99	16.02	0.42	0.06	2.90	-0.86
2010-2011	7.2	1.93	6.99	1.79	27.39	0.28	0.04	1.89	-1.27
≥1+ trout (<20 cm)									
2005-2009	4.0	0.84	0.60	0.13	2.71	1.41	0.22	8.85	0.34
2010-2011	2.4	0.65	1.16	0.26	5.17	0.57	0.09	3.55	-0.56
≥1+ trout (>20 cm)									
2005-2009	1.1	0.24	0.37	0.12	1.18	0.66	0.21	2.11	-0.41
2010-2011	0	0	0.40	0.12	1.28	0.67	0.21	2.17	-0.40

Table 4.8 HABSCORE outputs for site e (abstracted reach) before (2005-2009) and after (2010-2011) the Keltney Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2009	237.5	59.01	4.46	1.37	14.55	13.22	1.84	95.08	2.58
2010-2011	261.5	55.76	12.64	3.50	45.66	4.41	0.57	33.84	1.48
≥1+ salmon									
2005-2009	126.1	31.34	2.32	0.66	8.20	13.50	2.09	87.18	2.60
2010-2011	148.8	30.91	1.84	0.51	6.60	16.79	2.58	109.39	2.82
0+ trout									
2005-2009	12.5	3.09	10.61	2.73	41.19	0.29	0.04	1.98	-1.23
2010-2011	13.0	2.69	10.89	1.70	26.71	0.39	0.06	2.66	-0.9
≥1+ trout (<20 cm)									
2005-2009	7.4	1.85	3.62	0.81	16.05	0.51	0.08	3.17	-0.67
2010-2011	16.7	3.48	1.91	0.42	8.71	1.82	0.29	11.47	0.60
≥1+ trout (>20 cm)									
2005-2009	1.1	0.29	0.72	0.22	2.33	0.40	0.12	1.30	-0.91
2010-2011	1.4	0.29	0.71	0.22	2.35	0.41	0.12	1.37	-0.89

Table 4.9 HABSCORE outputs for site f (control) before (2005-2009) and after (2010-2011) the Keltney Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2009	109.2	22.19	6.88	1.94	24.35	3.23	0.43	24.47	1.17
2010-2011	159.8	31.06	3.95	1.16	13.50	7.86	1.06	58.34	2.06
≥1+ salmon									
2005-2009	26.3	5.34	1.42	0.39	5.18	3.77	0.57	24.94	1.32
2010-2011	55.9	10.86	1.78	0.48	6.52	6.11	0.92	40.53	1.81
0+ trout									
2005-2009	10.0	2.03	3.38	0.85	13.41	0.60	0.09	4.14	-0.51
2010-2011	11.2	2.18	5.40	1.35	21.60	0.40	0.06	2.81	-0.92
≥1+ trout (<20 cm)									
2005-2009	1.2	0.25	0.59	0.13	2.65	0.43	0.07	2.65	-0.84
2010-2011	3.9	0.75	0.95	0.21	4.21	0.79	0.13	4.91	-0.24
≥1+ trout (>20 cm)									
2005-2009	1.0	0.20	0.46	0.15	1.43	0.44	0.14	1.40	-0.80
2010-2011	0	0	0.21	0.07	0.65	0.93	0.30	2.91	-0.07

expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.7).

The observed density of 0+ salmon at site e prior to commissioning of the hydropower scheme (2005-2009 data) was higher than predicted by the Habitat Quality Score (HQS), suggesting better populations than expected, and HUI lower CL was >1 therefore the observed population was significantly higher than expected (Table 4.8). The observed density of 0+ salmon following commissioning of the hydropower scheme (2010-2011 data) was also higher than predicted (HQS) but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.8). The observed densities of ≥1+ salmon at site e prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were higher than predicted by the HQS, suggesting better populations than expected, and HUI lower CLs were >1 in both scenarios therefore the observed populations were significantly higher than expected (Table 4.8). The observed densities of 0+ trout at site e both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.8). The observed density of ≥1+ trout (<20 cm) at site e prior to commissioning of the hydropower scheme (2005-2009 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.8). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2010-2011 data), was higher than predicted (HQS), suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected. The observed densities of ≥1+ trout (>20 cm) at site e both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.8).

The observed density of 0+ salmon at site f prior to commissioning of the hydropower scheme (2005-2009 data) was higher than predicted by the HQS, suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.9). The observed density of 0+ salmon following commissioning of the hydropower scheme (2010-2011 data), was also higher than predicted by the HQS, suggesting better populations than expected, and the HUI lower CL was >1 therefore the observed population was significantly higher than expected (Table 4.9). The observed densities of 0+ trout at site f both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.9). The observed densities of 0+ trout (<20 cm) and 0+ trout (>20 cm) at site f both prior to and following commissioning of the hydropower scheme (2005-2009 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.9).

4.3.4 Salmonid growth rates

Upstream of the intake

At site a, upstream of the intake, back-calculated length of trout at age 1 ranged between 62 and 68 mm. Growth of trout in the first year of life was slowest in 2011, although this was based on observed mean length, and fastest in 2009 (Table 4.10). Back-calculated length of trout at age 2 was 116 mm in 2009.

Depleted reach

At site b, downstream of the intake, back-calculated length of trout at age 1 ranged between 53 and 82 mm. Growth of trout in the first year of life was slowest in 2006 and fastest in 2003 (Table 4.10). Back-calculated length of trout at age 2 ranged between 111 and 137 mm, with the slowest growth in 2004 and fastest in 2003 (Table 4.10). At site c, downstream of the intake, back-calculated length of trout at age 1 ranged between 52 and 65 mm. It should be noted that the growth calculations at sites a-c were based on small numbers of fish, often only one individual, so were subject to wide variation.

The sites in the lower reaches of Keltney Burn were situated close together and ultimately represent the same reach. Therefore ageing data from these three sites (d-f) were combined as fish are able to move freely along this reach. Back-calculated length of trout at age 1 ranged between 51 and 77 mm with observed length at age 1 in 2011 being 68 mm (Table 4.11). Growth of trout in the first year of life was slowest in 2003 and fastest in 2009 (Table 4.11). Back-calculated length of trout at age 2 ranged between 102 and 141 mm, with the slowest growth in 2003 and fastest in 2008 (Table 4.11). Back-calculated length of trout aged 3 ranged between 171 and 232 mm, with the slowest growth in 2004 and fastest in 2001 (Table 4.11). Back-calculated length of trout aged 4 ranged between 239 and 355 mm (Table 4.11). The last few year groups were based on small numbers of fish, often only one individual so are subject to high variation.

Table 4.10 Back-calculated lengths at age of trout at sites a, b and c in the upper reaches of Keltney Burn. Note no 0+ trout were captured in 2011 at site b and c. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))		
		1	2	3
a	2009	68(1)	116(1)	
	2010	65±7(5)		
	*2011	62±6(3)		
b	2003	82(1)	137(1)	
	2004	75±6(2)	111(1)	
	2005	62±8(3)	117±19(2)	
	2006	53±6(6)		
	2009	74(1)		
c	2006	65±13(9)		
	2008	52(1)	104(1)	157(1)

Table 4.11 Back-calculated lengths at age of trout at sites d, e and f in the lower reaches of Keltney Burn. * see Section 3.3.4 for details.

Year class	Back-calculated length at age (mm ± S.D (n))				
	1	2	3	4	5
2001	70±14(2)	136±21(2)	232±23(2)	355±12(2)	403±17(2)
2002	60(1)	120(1)	194(1)	239(1)	
2003	51(1)	102(1)			
2004	65±13(32)	136±16(10)	171±18(2)		
2005	63±8(6)	133±21(2)			
2006	64±11(29)	109±11(2)	186(1)		
2007	52±3(5)	111±11(3)			
2008	71±10(5)	141±0(2)	203(1)		
2009	77±11(28)	119±10(6)			
2010	73±10(14)				
*2011	68±6(16)				

Back-calculated length of salmon at age 1 ranged between 39 and 67 mm with observed length at age 1 in 2011 being 56 mm. Growth of salmon in the first year of life was slowest in 2003 and fastest in 2009 and 2010 (Table 4.12). Back-calculated length of salmon at age 2 ranged between 74 and 108 mm, with the slowest growth in 2002 and fastest in 2008 (Table 4.12). Back-calculated length of salmon aged 3 and 4 was 123 mm and 160 mm respectively (Table 4.12). The last two year groups were based on small numbers of fish, often only one individual so were subject to high variation.

Overall during the study period the growth of trout and salmon within the abstracted reaches followed similar trends to the growth of trout and salmon at sites outside the abstracted reach.

Table 4.12 Back-calculated lengths at age of salmon at sites d, e and f in the lower reaches of Keltney Burn. * see Section 3.3.4 for details.

Year class	Back-calculated length at age (mm ± S.D (n))				
	1	2	3	4	5
2002	43(1)	74(1)	123(1)	160(1)	
2003	39±5(8)	89±6(8)			
2004	48±7(76)	93±8(47)			
2005	60±6(17)	95(1)			
2006	52±7(17)	90±6(2)			
2007	56±10(17)	80±5(4)			
2008	53±10(15)	108±10(5)			
2009	67±10(61)				
2010	67±13(19)				
*2011	56± 6(398)				

4.3.5 Eel

Numbers of eels captured at sites d-f varied between survey years and data suggest the lower reaches of Keltney Burn are used by small numbers of eels (Figure 4.16, Table 4.13).

Table 4.13 Number of eel captured at survey sites in the lower reaches of Keltney Burn between 2005 and 2011.

Year	Site identifier		
	d	e	f
2005	12	15	7
2006	4	4	14
2007	5	16	11
2008	7	14	4
2009	4	14	2
2010	6	48	9
2011	2	26	8

At site d and f numbers of eels caught over the study period were similar pre and post hydropower commissioning; at site e numbers of eels caught post hydropower commissioning were higher than those caught prior to hydropower commissioning (Table 4.13). Despite variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers with the hydropower scheme operational.

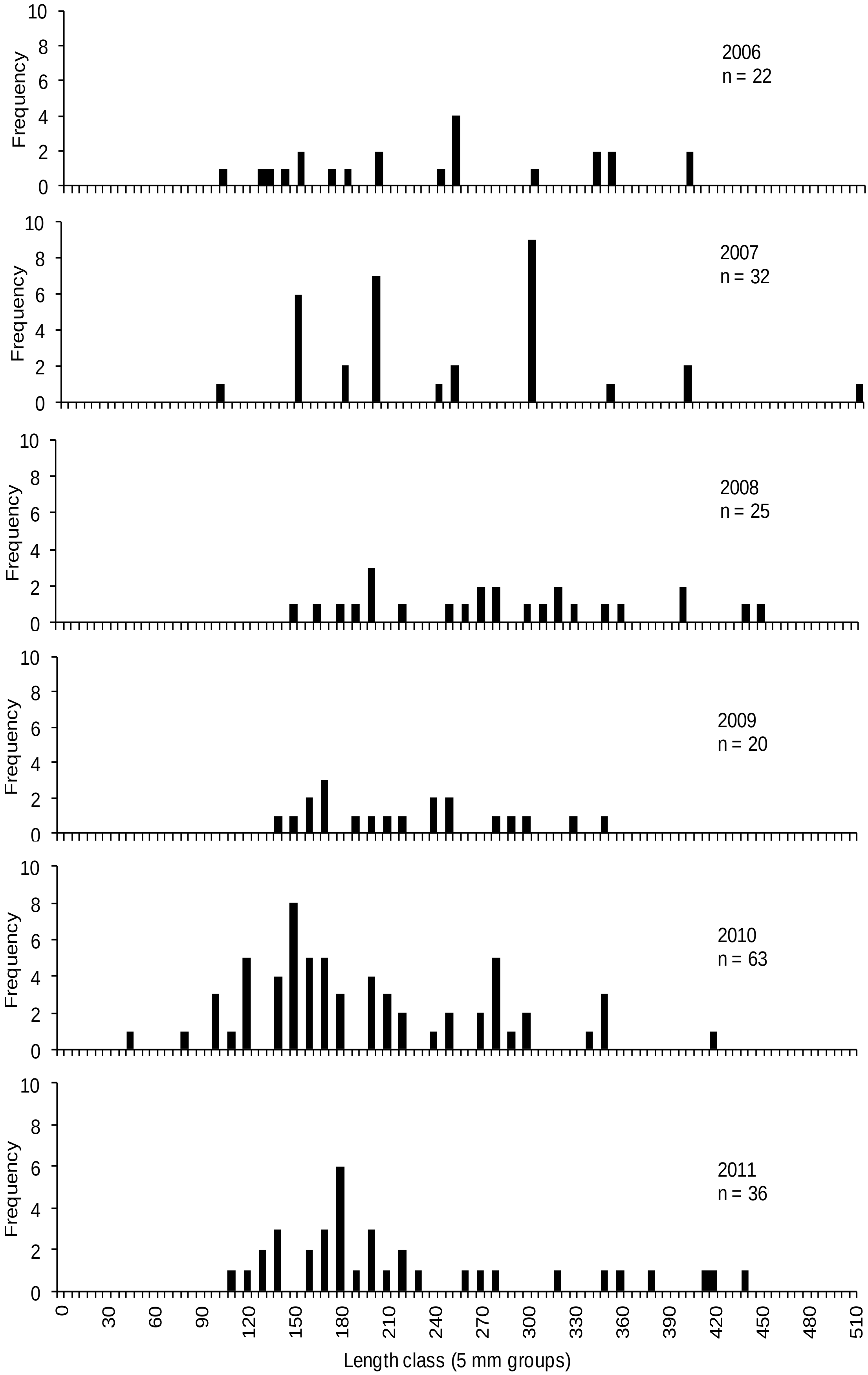

Figure 4.16 Combined length distributions of eels at sites d-f in Keltney Burn between 2006 and 2011. Note eel raw lengths in 2005 were not recorded.

4.3.6 Wetted widths

A paired two sample *t*-test was performed on mean wetted widths (collected during HABSCORE) at site d and e, as these were located within the abstracted reach. Essentially, the test was used to distinguish if there was any difference between wetted widths each time HABSCORE was performed to ensure comparison of density estimates over the study period. At site d the 95% confidence limits (-1.492, 0.225) enclosed zero with a p value of 0.139, n = 42. At site e the 95% confidence limits (-2.593, 0.005) enclosed zero with a p value of 0.051, n = 42; therefore there was no significant difference between wetted widths collected during the surveys pre and post abstraction, confirming comparability of density estimates over the study period.

4.4 Rottal Burn hydropower scheme

4.4.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

No salmon were captured in the upper reaches associated with the intakes of Rottal Burn throughout the study period.

Upstream of the intakes

0+ trout densities at site a, upstream of the intake on Burn of Heughs, varied between study years with the greatest density in 2009, classified as excellent (class A), and the lowest density in 2011, classified as poor (class E) (Figure 4.17, Table 4.14). 0+ trout populations in 2006 and 2007 were similar and good (class B), decreasing to fair/average (class C) in 2010 and poor (class E) in 2011. ≥1+ trout densities were similar in all study years except 2011 when densities were lower; populations were classified as excellent (class A) in all years (Figure 4.17, Table 4.14).

0+ trout densities at site c, upstream of the intake on Kennel Burn, varied between study years with similar densities in 2006, 2009 and 2011 but an absence of 0+ trout in 2007 and 2010 (Figure 4.17, Table 4.14); populations ranged between fair/average (class C) and absent (class F). ≥1+ trout densities varied over the study period, with the greatest density in 2006 and the lowest in 2009; despite this variation populations were excellent (class A) in all years except in 2009 (Figure 4.17, Table 4.14).

Table 4.14 Classification of trout densities in both the upper and lower reaches of Rottal Burn between 2005 and 2011 based on five classification schemes. Note hydropower scheme commissioned December 2008. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2006	B	C	B	D	B
	2007	B	C	C	E	B
	2009	A	B	B	C	B
	2010	C	C	C	E	B
	2011	E	E	E	E	E
b*	2006	D	E	E	E	D
	2007	D	E	E	E	D
	2009	C	D	D	E	C
	2010	B	C	C	E	B
	2011	D	E	E	E	D
c	2006	C	D	D	E	C
	2007	F	F	F	F	F
	2009	C	C	C	E	C
	2010	F	F	F	F	F
	2011	C	D	D	E	C
d*	2006	B	C	C	E	B
	2007	D	D	E	E	D
	2009	B	C	B	D	B
	2010	B	C	C	D	B
	2011	B	C	B	D	A
e*	2006	n/s	n/s	n/s	n/s	n/s
	2007	F	F	F	F	F
	2009	E	E	E	E	E
	2010	D	E	E	E	D
	2011	C	D	D	E	C
f*	2006	D	E	C	D	D
	2007	E	E	E	E	E
	2009	D	E	D	E	D
	2010	C	D	D	E	C
	2011	D	E	D	E	D
g	2006	D	E	E	E	D
	2007	F	F	F	F	F
	2009	D	D	D	E	D
	2010	D	E	E	E	D
	2011	D	D	C	D	D

≥1+ trout

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2006	A	A	A	A	A
	2007	A	A	A	A	A
	2009	A	A	A	A	A
	2010	A	A	A	A	A
	2011	A	B	A	B	B
b*	2006	A	A	A	A	A
	2007	A	A	A	A	A
	2009	A	A	A	B	A
	2010	A	A	A	A	A
	2011	A	A	A	B	A
c	2006	A	A	A	A	A
	2007	A	A	A	A	A
	2009	B	B	B	C	B
	2010	A	A	A	A	A
	2011	A	B	B	C	B
d*	2006	A	A	A	A	A
	2007	A	A	A	A	A
	2009	A	B	B	C	B
	2010	A	A	A	A	A
	2011	A	A	A	B	A
e*	2006	n/s	n/s	n/s	n/s	n/s
	2007	B	C	A	B	C
	2009	A	A	A	A	A
	2010	B	B	B	C	B
	2011	A	B	B	C	B
f*	2006	A	A	A	A	A
	2007	D	D	D	D	D
	2009	B	C	A	B	B
	2010	B	C	B	B	B
	2011	D	D	C	D	D
g	2006	D	D	D	E	C
	2007	D	D	E	E	C
	2009	C	C	C	D	C
	2010	E	E	E	E	C
	2011	E	E	E	E	E

Depleted reach

At site b, downstream of the intake on Burn of Heughs, 0+ trout densities were generally lower in all years than at site a except in 2010 (Figure 4.17). In 2006 and 2007 densities of 0+ trout were fair/poor (class D), improving in 2009 and 2010 to fair/average (class C) and good (class B) respectively; 0+ trout populations decreased in 2011 to similar densities observed in 2006 and 2007. ≥1+ trout densities varied over the study period with the greatest density in 2006 and the lowest in 2009; despite this variation populations were excellent (class A) in all years (Figure 4.17, Table 4.14).

At site d, downstream of the intake on Kennel Burn, 0+ trout densities were higher than at site c in all corresponding years (Figure 4.17). 0+ trout populations were good (class B) in all years except 2007 when populations were fair/poor (class D); 0+ trout populations were higher between 2009 and 2011 than 2006 and 2007. ≥1+ trout densities at site d varied over the study period with the greatest density in 2006 and the lowest in 2009; despite this variation populations were excellent (class A) in all years (Figure 4.17, Table 4.14).

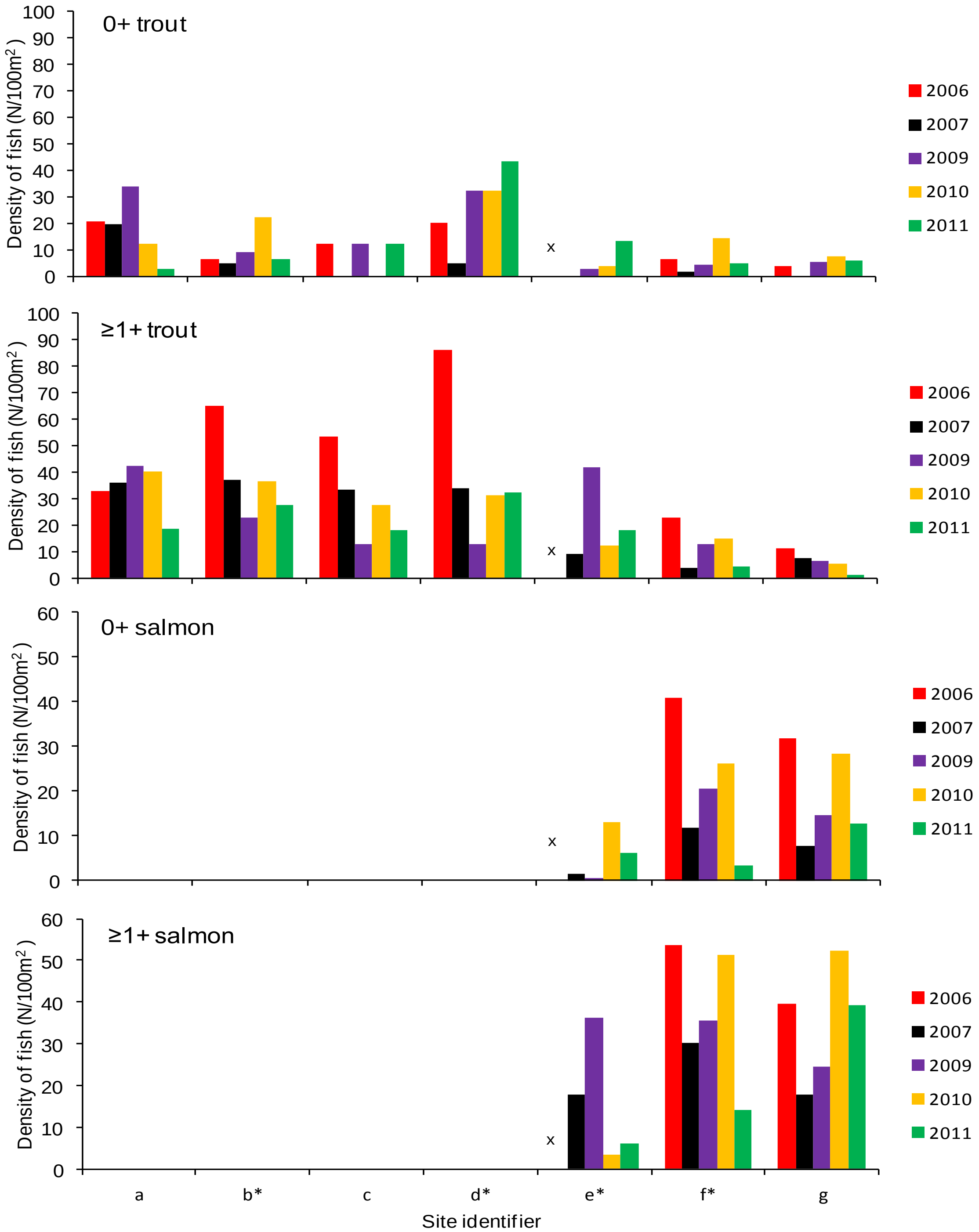

Figure 4.17 Density estimates of trout and salmon in Rottal Burn between 2006 and 2011. Note no sites were sampled in 2008. Hydropower scheme was commissioned in December 2008. x represents years not surveyed.

Site e was located in the middle reaches of the abstracted reach in Burn of Heughs and surveyed to assess salmon penetration beyond the outfall; the site was not surveyed in 2006. 0+ salmon densities were low in 2007 and 2009 with the highest densities found in 2010 and 2011; populations were poor (class E) throughout the study period except in 2010 when populations were fair/average (class C) (Figure 4.17, Table 4.15). ≥1+ salmon densities varied over the study period with the greatest density in 2009 and the lowest in 2010; populations were good (class B) in 2007 and 2009 but fair/poor (class D) in 2010 and 2011 (Figure 4.17, Table 4.15).

Table 4.15 Classification of salmon densities in the lower reaches of Rottal Burn between 2005 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
e*	2006	n/s	n/s	n/s	n/s	n/s
	2007	E	E	E	E	E
	2009	E	E	E	E	E
	2010	C	D	C	D	D
	2011	E	E	E	E	E
f*	2006	C	D	C	D	C
	2007	D	D	D	E	D
	2009	C	D	D	D	D
	2010	C	D	D	D	C
	2011	E	E	E	E	E
g	2006	C	D	C	D	C
	2007	E	E	E	D	E
	2009	E	E	E	E	D
	2010	D	D	D	D	C
	2011	D	D	D	E	D

≥1+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
e*	2006	n/s	n/s	n/s	n/s	n/s
	2007	B	B	B	B	B
	2009	B	A	A	A	A
	2010	D	D	D	D	D
	2011	D	D	D	D	C
f*	2006	A	A	A	A	A
	2007	B	B	B	B	A
	2009	A	B	B	B	A
	2010	A	A	A	A	A
	2011	C	C	C	D	B
g	2006	A	A	A	A	A
	2007	C	C	C	C	B
	2009	B	B	B	B	A
	2010	B	B	B	B	A
	2011	B	B	B	C	A

0+ trout densities at site e varied between years with 0+ trout absent in 2007 but populations increased between 2009 and 2011; 0+ trout populations were poor (class E) in 2009 increasing to fair/average (class C) by 2011 (Figure 4.17, Table 4.14). ≥1+ trout populations were highest in 2009 and lowest in 2007, and populations varied between excellent (class A) and good (class B) during the study period (Figure 4.17, Table 4.14).

At site f, upstream of the outfall, 0+ salmon populations were highest in 2006 and lowest in 2011 ranging from fair/average (class C) to poor (class E), following a similar trend to populations at site g, downstream of the outfall (Figure 4.17, Table 4.15). ≥1+ salmon populations ranged between excellent (class A) and fair/average (class C) during the study period with the lowest densities in 2011, following a similar trend to populations at site g (Figure 4.17, Table 4.15).

At site f 0+ trout populations were lower than salmon with the highest densities in 2010 and lowest in 2007 ranging from fair/average (class C) to poor (class E), following a similar trend to populations at site g (Figure 4.17, Table 4.14). ≥1+ trout populations were also lower than salmon with the highest densities in 2006 and lowest in 2007 ranging from excellent (class A) in 2006 to fair/average (class D) in 2007 and 2011; populations were higher than at site g except in 2007 (Figure 4.17, Table 4.14).

Downstream of the outfall

At site g, downstream of the outfall, 0+ salmon populations were highest in 2006 and lowest in 2007 with a similar trend to those found at site f (Figure 4.17). 0+ salmon populations were fair/average (class C) in 2006, poor (class E) in 2007 and 2009 improving to fair/poor (class D) in 2010 and 2011 (Table 4.15). ≥1+ salmon populations were highest in 2010 and lowest in 2007 with a similar trend to those found at site f (Figure 4.17). ≥1+ salmon populations were excellent (class A) in 2006, fair/average (class C) in 2007 before improving to good (class B) in subsequent years (Table 4.15).

At site g, 0+ trout populations were low throughout the study period as found at sites f and g (Figure 4.17). 0+ trout populations were fair/poor (class D) in all years except 2007 when no 0+ trout were captured (Table 4.14). ≥1+ trout populations were highest in 2006 and lowest in 2011 with a similar trend to those found at site f (Figure 4.17). ≥1+ trout populations were fair/poor (class D) in 2006 and 2007 decreasing to poor (class E) in 2010 and 2011 (Table 4.14).

4.4.2 Salmonid size structure

Upstream of the intakes

Length distributions of trout in the upper reaches associated with the two intakes varied between years at each site.

At site a, upstream of the intake on Burn of Heughs, 0+ trout were caught in good numbers in the size range 40-80 mm throughout the study period except in 2011 when only three 0+ trout were caught in the size range 55-70 (Figure 4.18). ≥1+ trout in 2006 were caught in a narrow size range of 85-140 mm, while in other years ≥1+ trout were caught in a wider size range of 75-180 mm albeit in low numbers in 2011 (Figure 4.18); one larger individual >180 mm was captured in 2011.

At site c, upstream of the intake on Kennel Burn, 0+ trout were absent in 2007 and 2010 and caught in the size range 45-75 mm in other years (Figure 4.20). ≥1+ trout were caught in a similar size range (85-180 mm) throughout the study period albeit in variable numbers, except in 2007 when one larger individual > 180 mm was captured (Figure 4.20).

Depleted reach

At site b, downstream of the intake on Burn of Heughs, 0+ trout were caught in the size range 40-75 mm in 2009 and 2010 with a marginally narrower size range in other years (Figure 4.19). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2006, 2010 and 2011 when a small number of larger individuals >180 mm were captured (Figure 4.19). The findings reflect the size structure found at site a upstream of the intake on the Burn of Heughs.

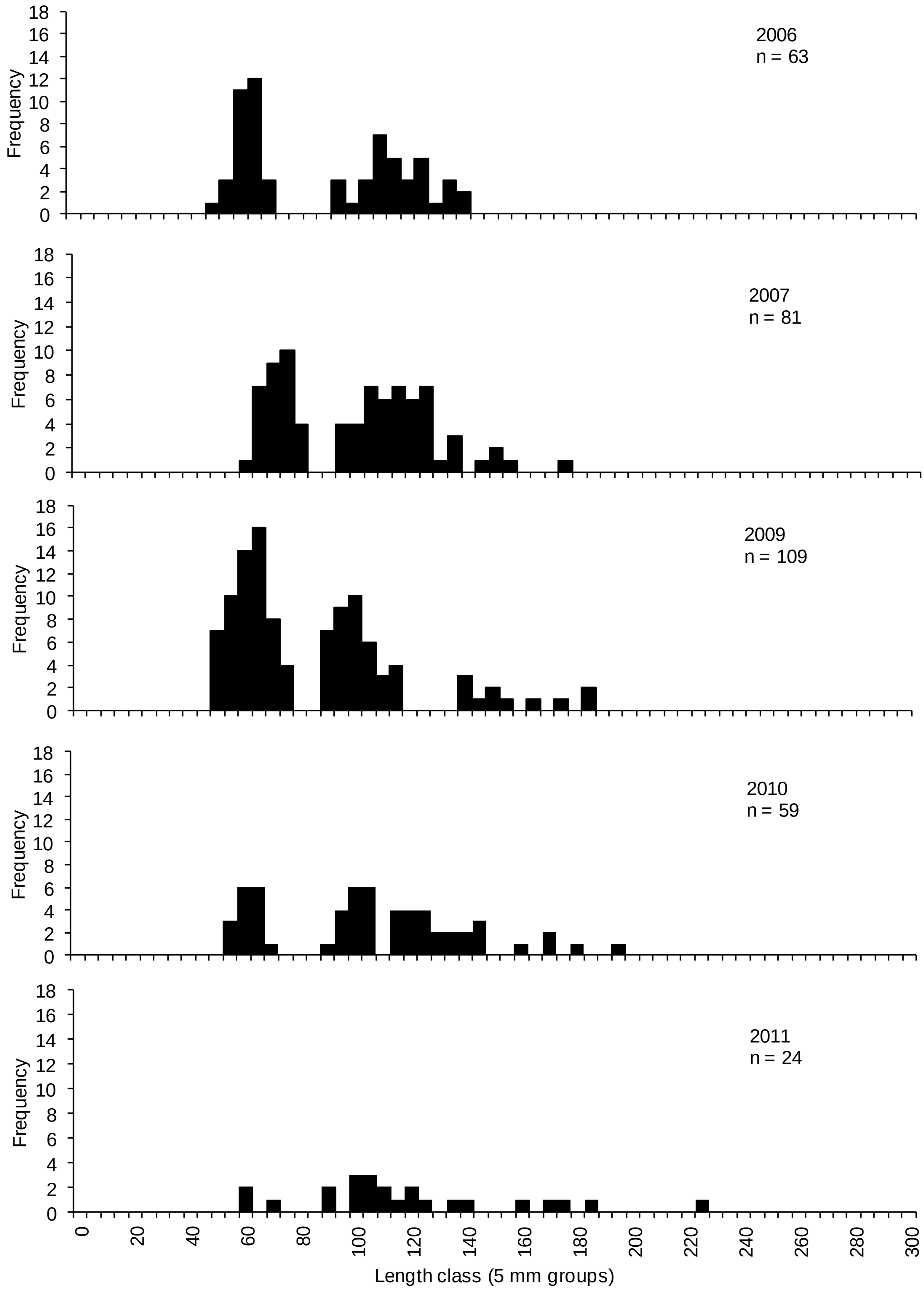

Figure 4.18 Length distributions of trout at site a in Rottal Burn between 2006 and 2011. Note site a was not surveyed in 2008.

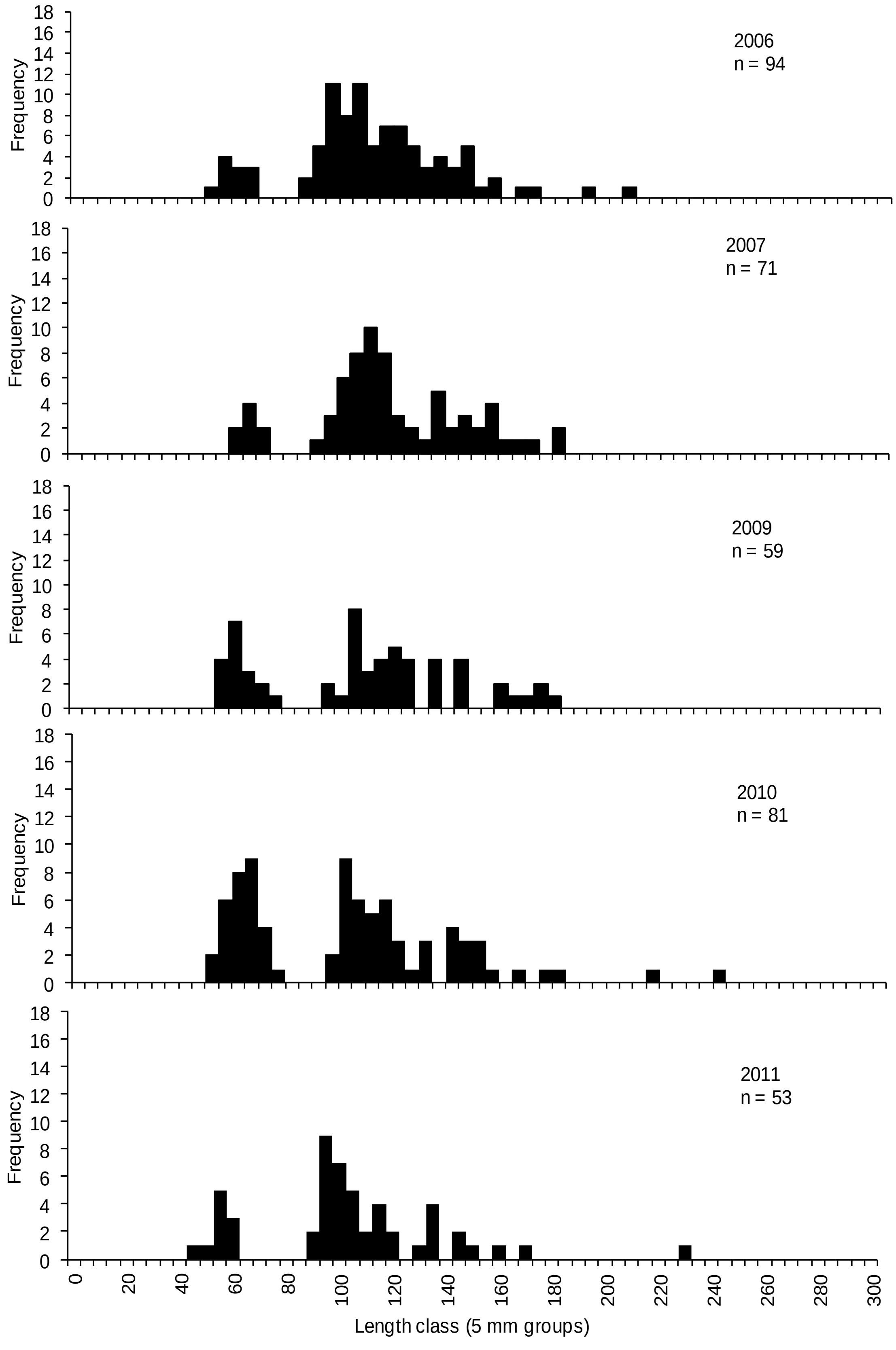

Figure 4.19 Length distributions of trout at site b in Rottal Burn between 2006 and 2011. Note site b was not surveyed in 2008.

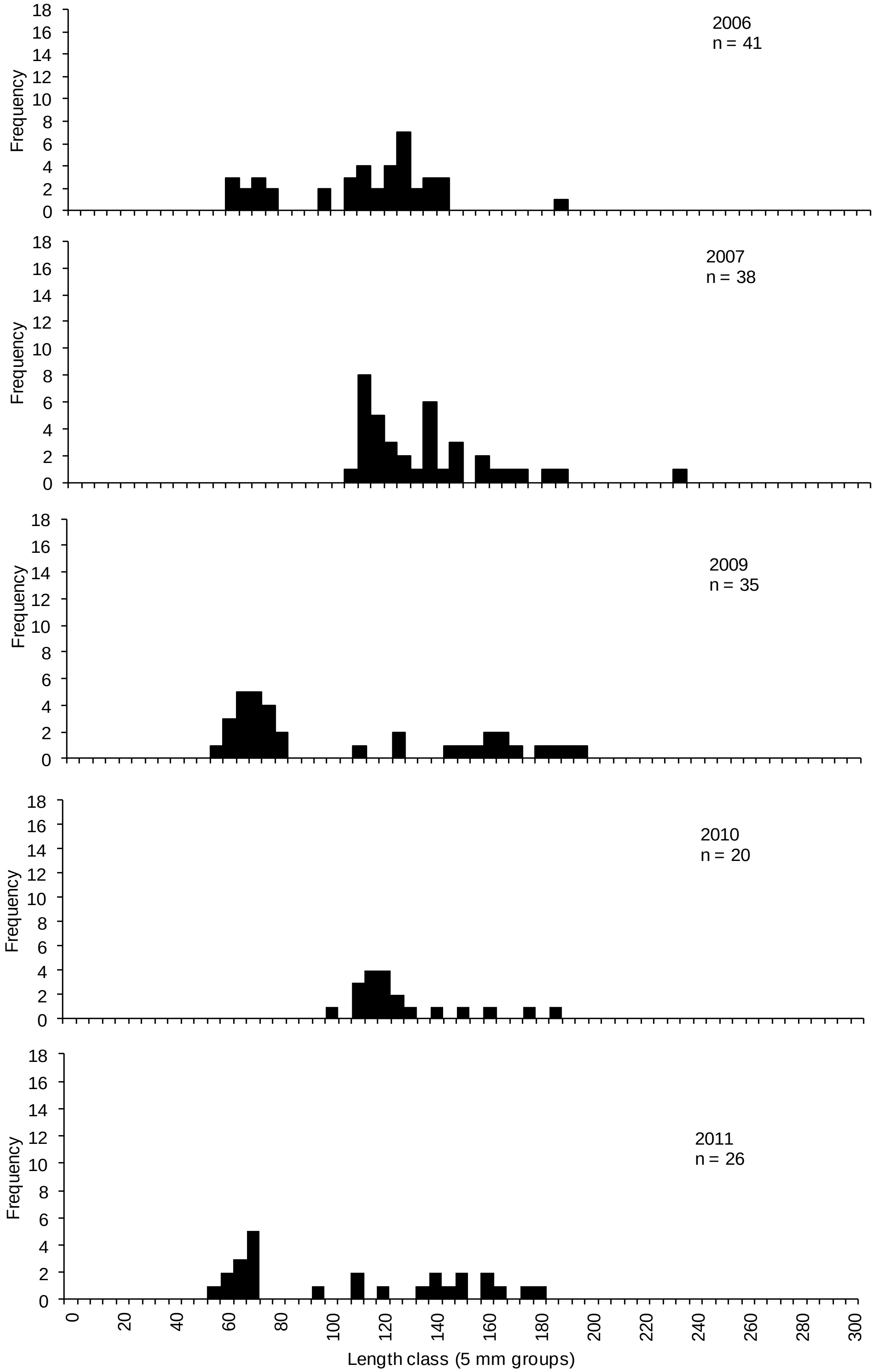

Figure 4.20 Length distributions of trout at site c in Rottal Burn between 2006 and 2011. Note site c was not surveyed in 2008.

At site d, downstream of the intake on Kennel Burn, 0+ trout were caught in a similar size range to site c, namely 45-70 mm except in 2007 when they were caught in a narrower size range (Figure 4.21). ≥1+ trout were caught in a similar size range (80-180 mm) throughout the study period except in 2006, 2007 and 2011 when a small number of larger individuals >180 mm were captured (Figure 4.21). The findings reflect the size structure found at site c upstream of the intake on Kennel Burn.

In the middle reaches of Rottal Burn at site e within the abstracted reach of Burn of Heughs, 0+ salmon were caught in the size range 45-65 mm in 2010 while in other years the size range was narrower (Figure 4.22). ≥1+ salmon were caught in a similar size range (80-130 mm) in 2007 and 2009, with one fish >130 mm in the latter year; in 2010 and 2011 ≥1+ salmon were caught in a narrower size range of 85-120 mm, with one salmon >130 mm caught in 2010 (Figure 4.22). 0+ trout were absent in 2007 and caught in the size range of 40-70 mm in other years (Figure 4.23). ≥1+ trout were caught in wide size range (80-220 mm) in 2009 but a narrow size range in other years (Figure 4.23).

Length distributions of salmon in the lower reaches of Rottal Burn were similar between years, albeit in varying numbers, at both sites f and g (Figures 4.27 & 4.28). At site f, upstream of the outfall, 0+ salmon were caught in the size range 35-75 mm while ≥1+ salmon were caught in the size range 80-130 mm, throughout the study period except in 2010 when a salmon of 167 mm was caught (Figure 4.24). At site f 0+ trout were caught in the size range 40-75 mm, albeit in variable numbers, in all study years except 2007 when they were caught in a narrower size range (Figure 4.26). ≥1+ trout were caught in wide size range (80-180 mm) in 2006 but a narrow size range in other years (Figure 4.26).

Downstream of the outfall

At site g, downstream of the outfall, 0+ salmon were caught in the size range 40-70 mm while ≥1+ salmon were caught in the size range 75-125 mm, throughout the study period except in 2011 when a salmon of 149 mm was caught (Figure 4.25). 0+ trout were absent in 2007 and caught in the size range 45-75 mm in other years (Figure 4.27). ≥1+ trout were caught in wide size range (80-200 mm) in 2007 but a narrow size range in other years (Figure 4.27).

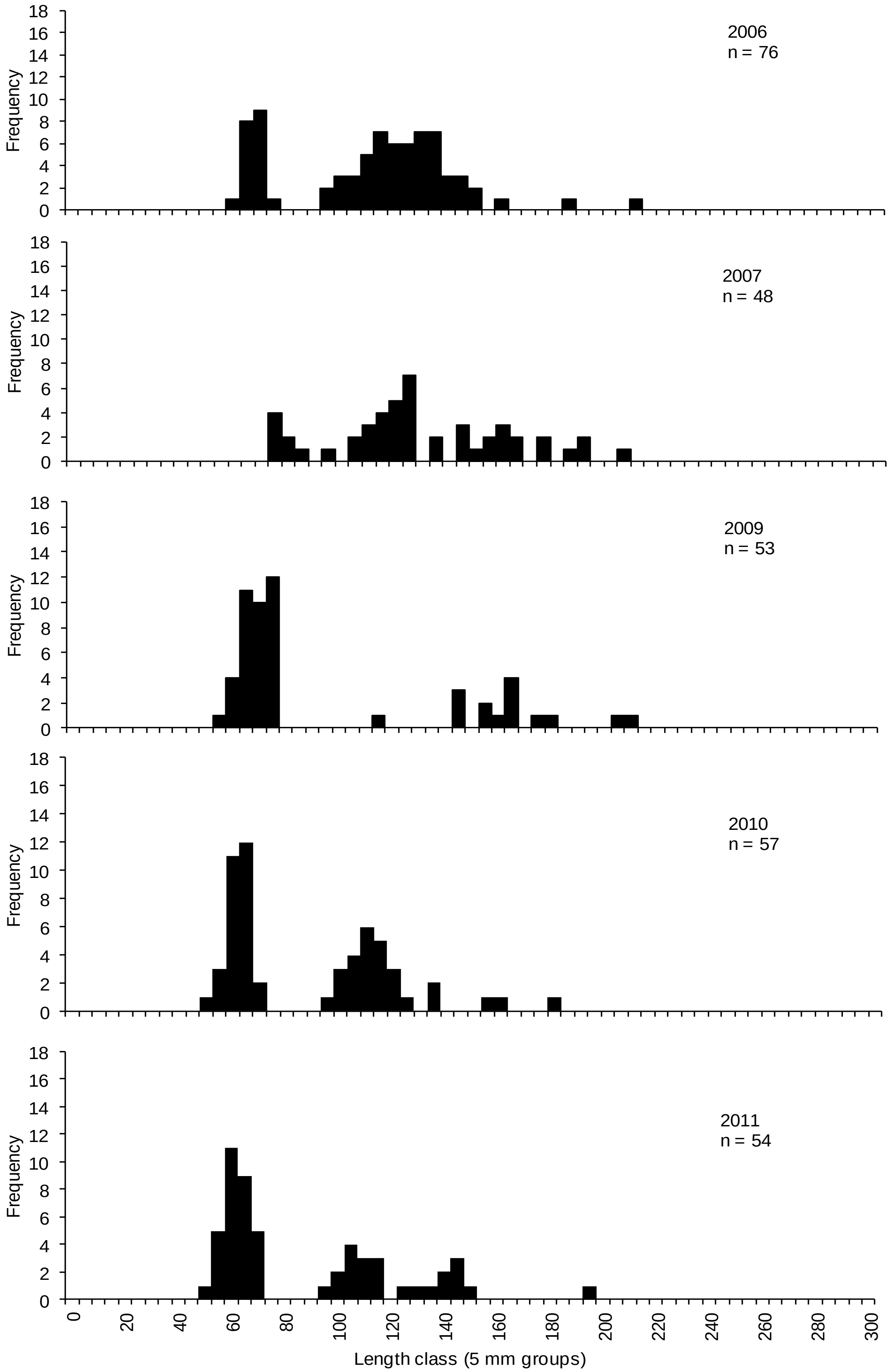

Figure 4.21 Length distributions of trout at site d in Rottal Burn between 2006 and 2011. Note site d was not surveyed in 2008.

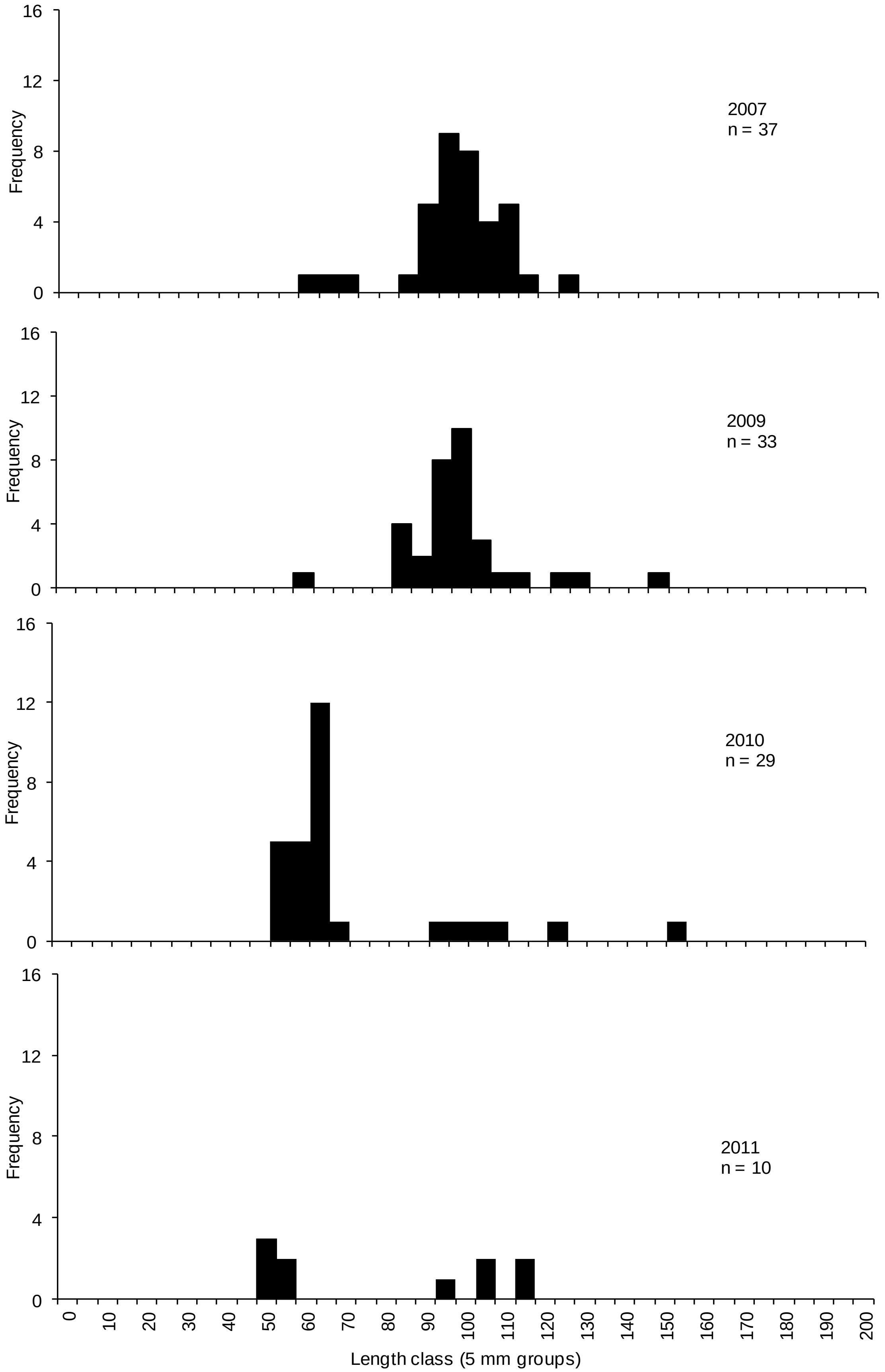

Figure 4.22 Length distributions of salmon at site e in Rottal Burn between 2007 and 2011. Note site e was not surveyed in 2006 or 2008.

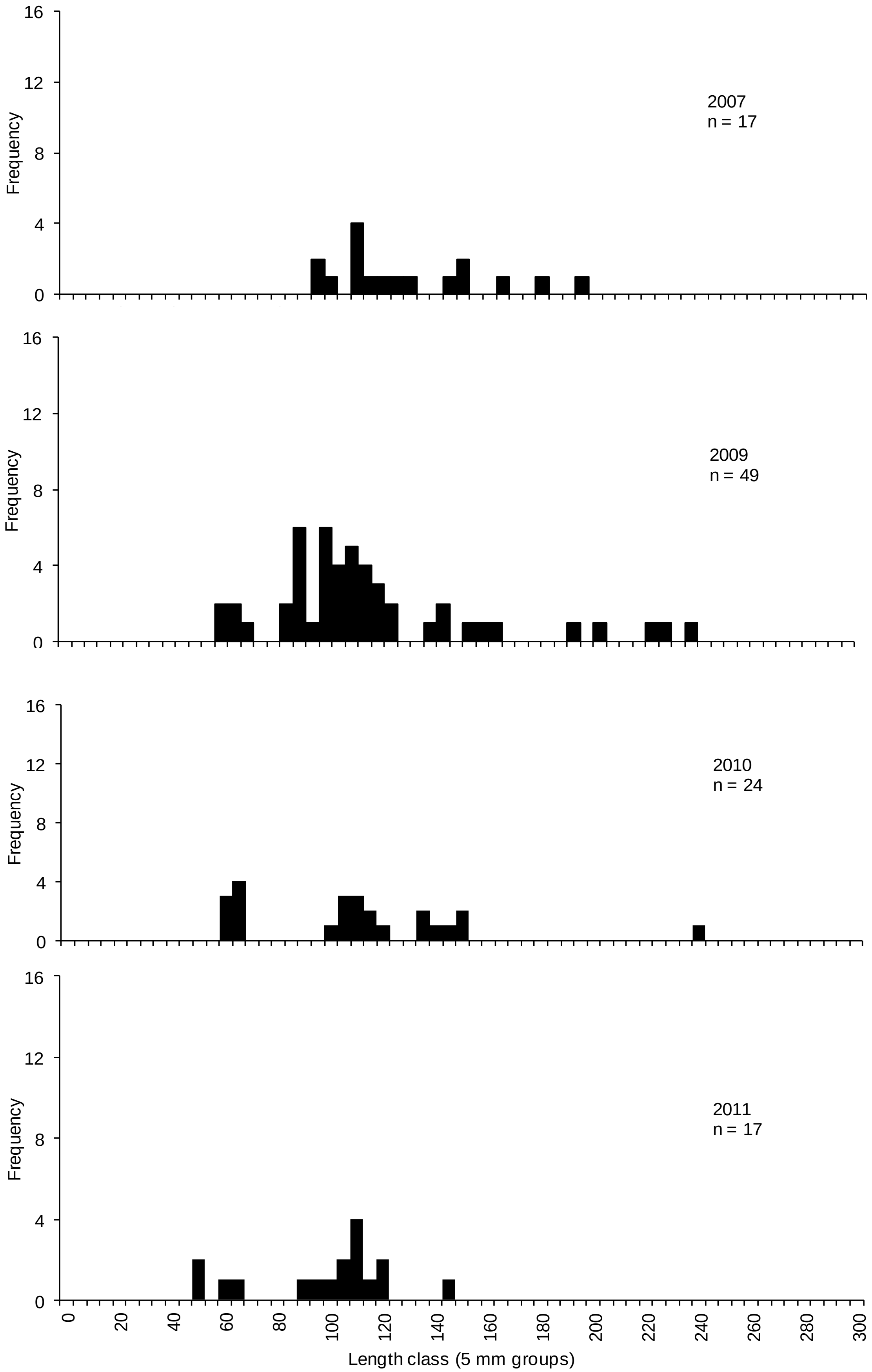

Figure 4.23 Length distributions of trout at site e in Rottal Burn between 2007 and 2011. Note site e was not surveyed in 2006 and 2008.

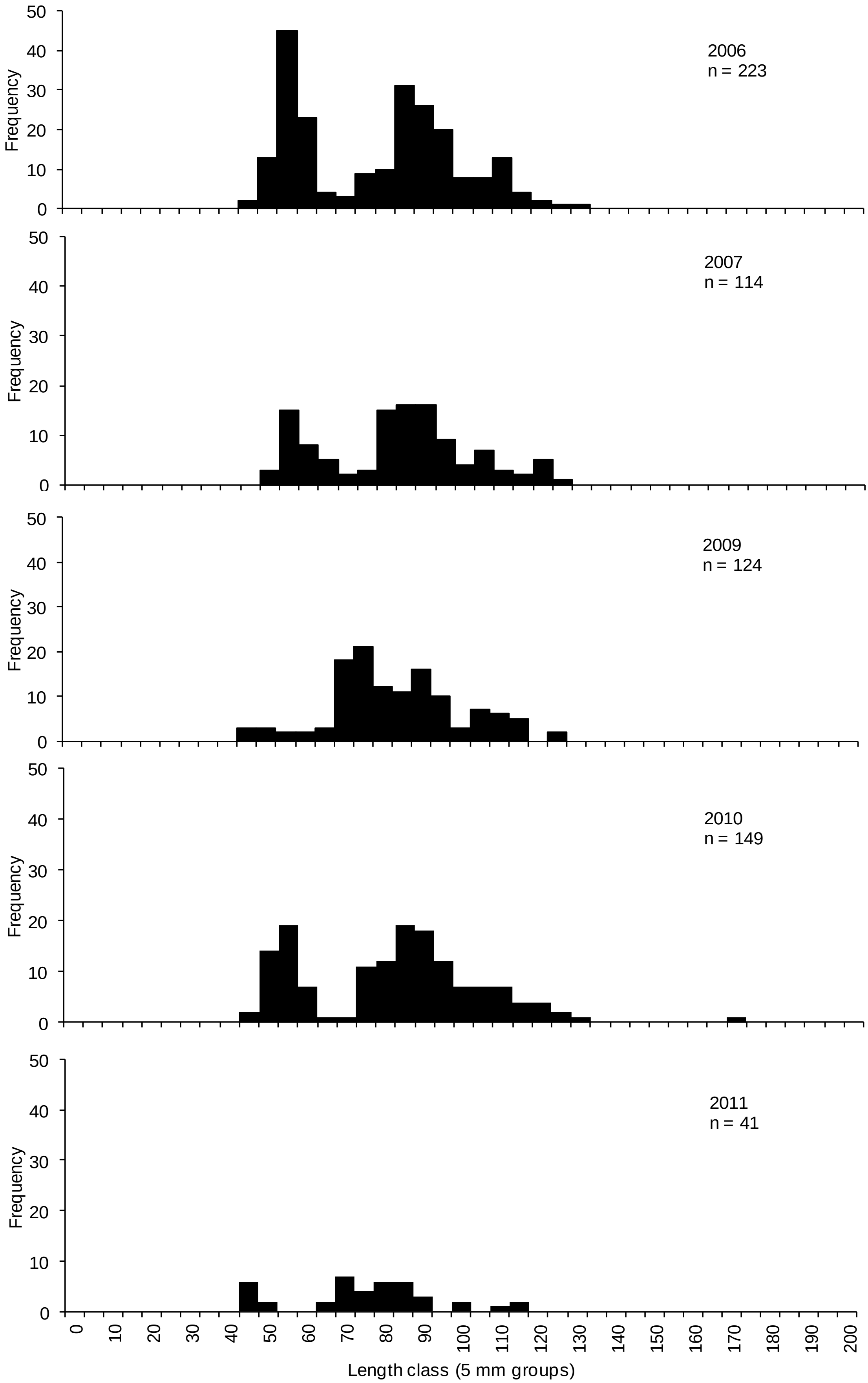

Figure 4.24 Length distributions of salmon at site f in Rottal Burn between 2006 and 2011. Note site f was not surveyed in 2008.

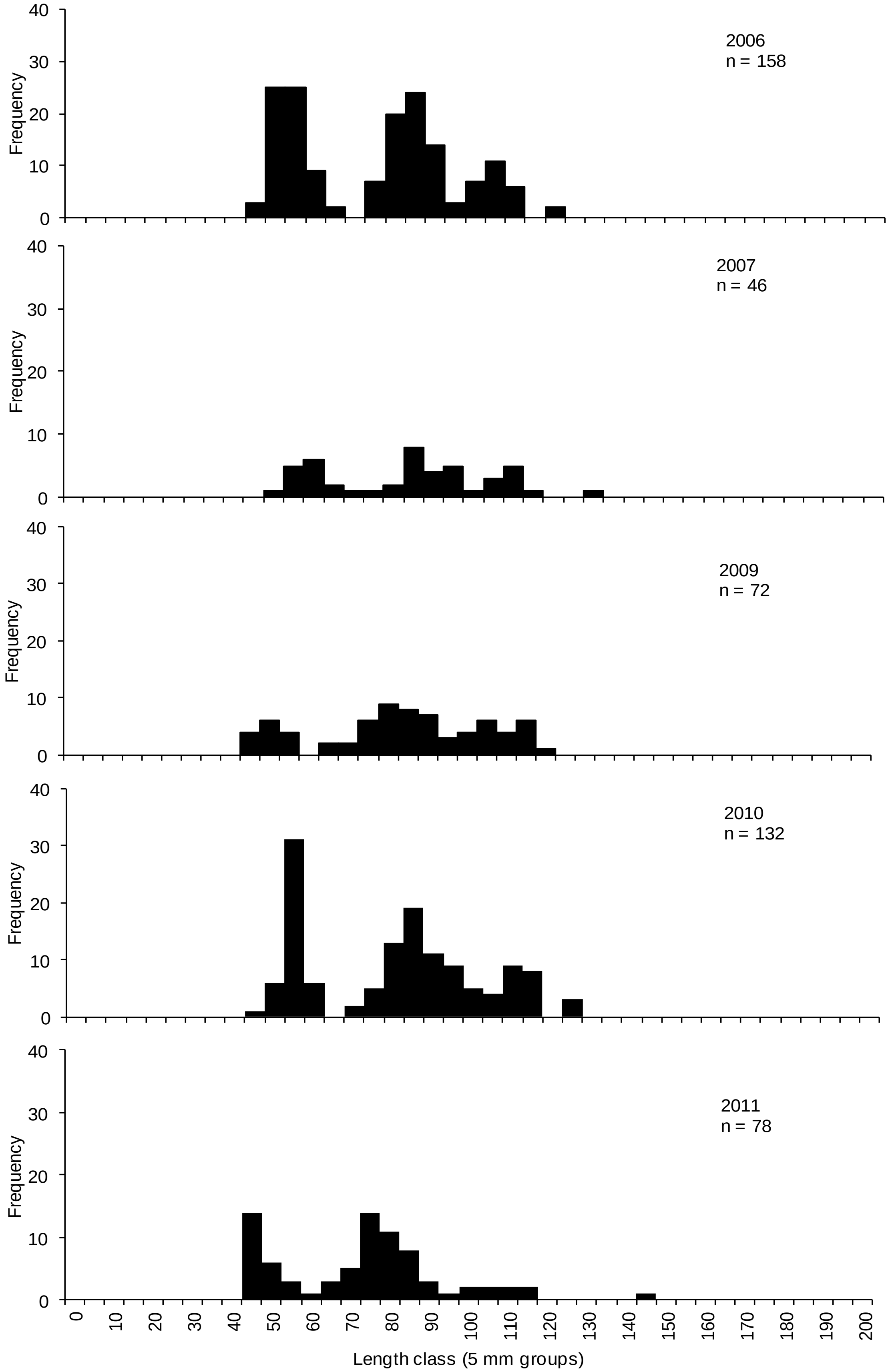

Figure 4.25 Length distributions of salmon at site g in Rottal Burn between 2006 and 2011. Note site g was not surveyed in 2008.

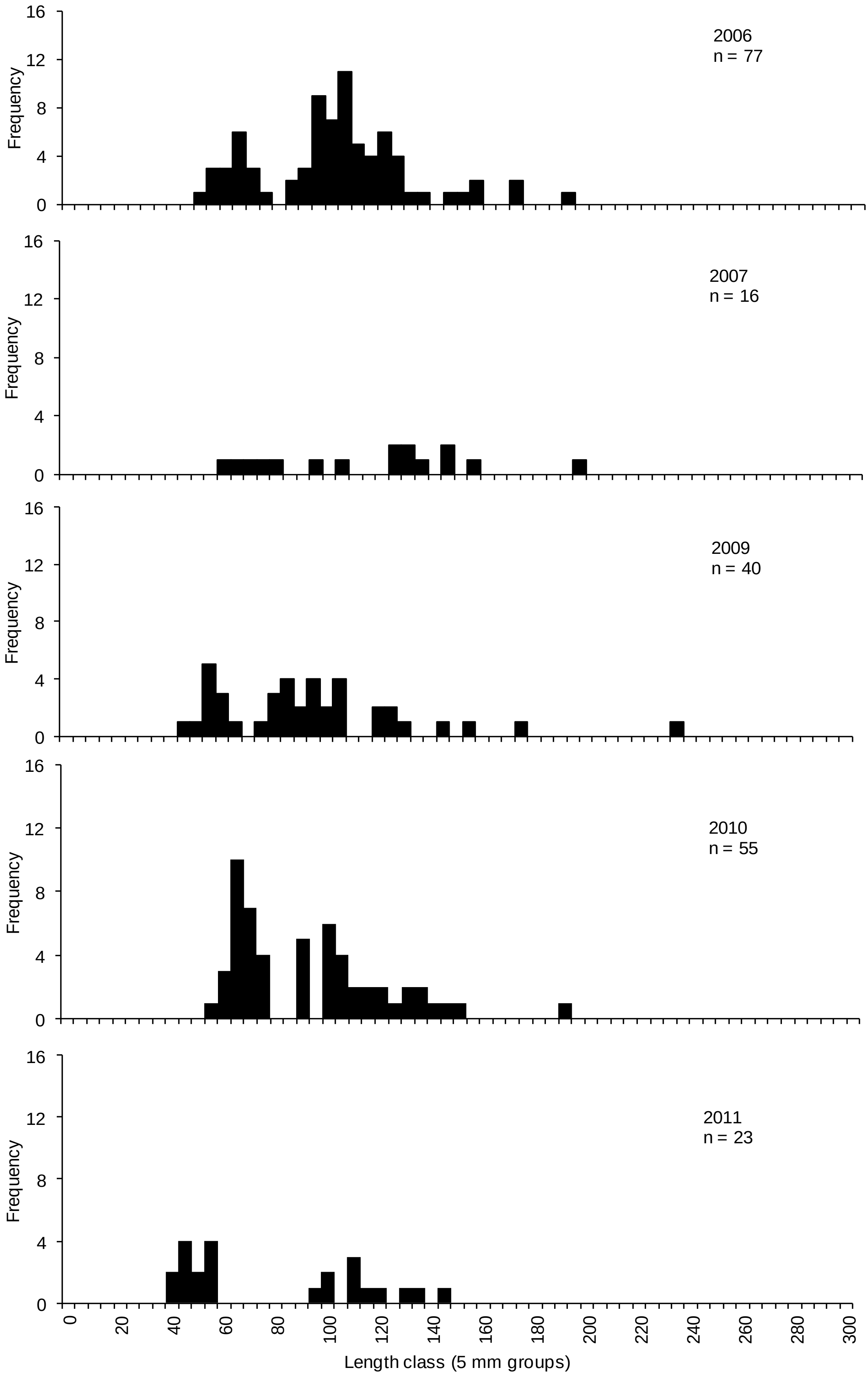

Figure 4.26 Length distributions of trout at site f in Rottal Burn between 2006 and 2011. Note site f was not surveyed in 2008.

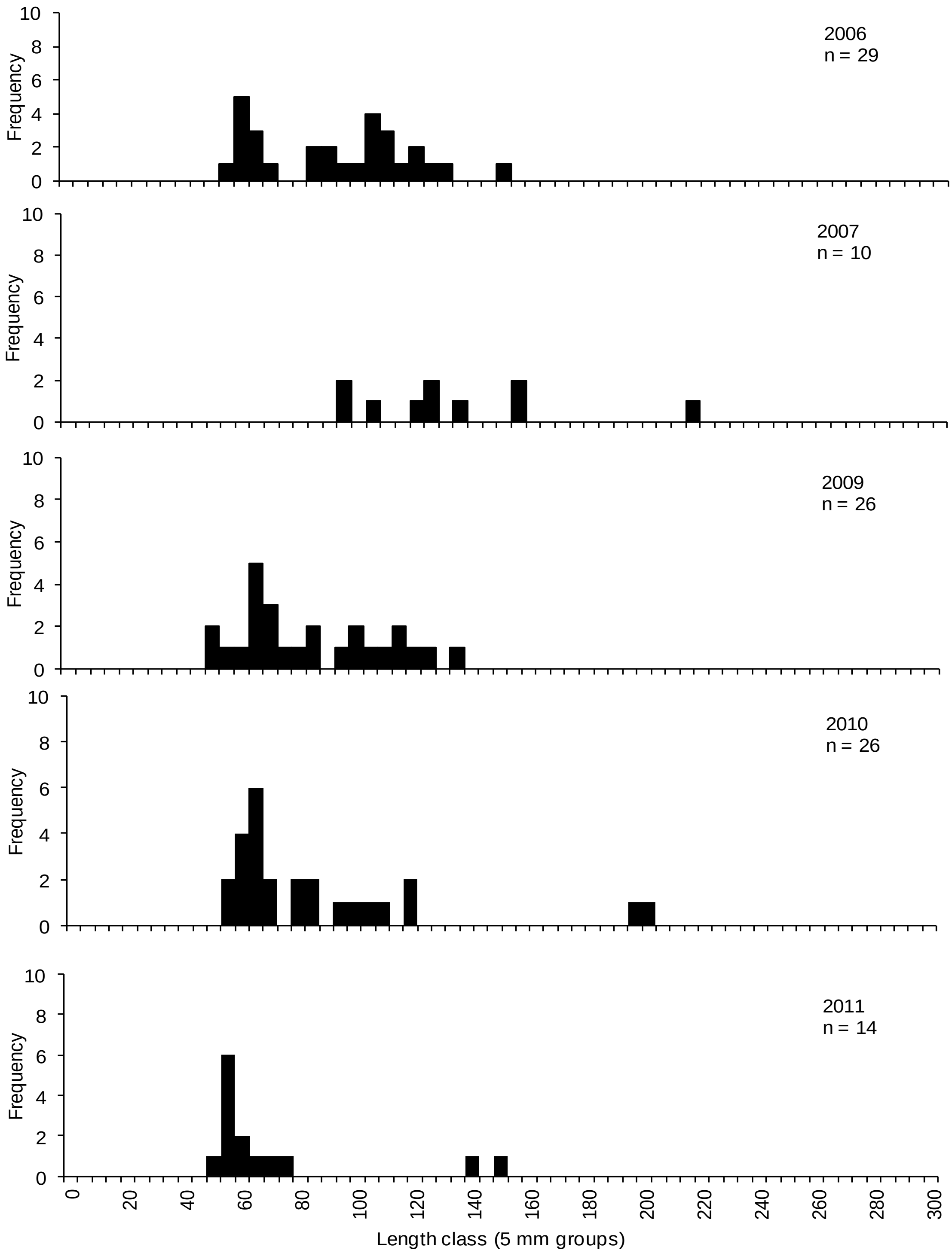

Figure 4.27 Length distributions of trout at site g in Rottal Burn between 2006 and 2011. Note site g was not surveyed in 2008.

4.4.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was commissioned were compared to the same information collected following commissioning at two key sites in the lower reaches of Rottal Burn. For the analysis habitat data collected in 2007 were utilised with fisheries data collected from 2006-2007 and compared to habitat data collected in 2011 utilised with fisheries data collected from 2009-2011 (Tables 4.16 & 4.17). The data allowed comparison between one site in the abstracted reach and one site downstream of the abstracted reach. The observed densities of 0+ salmon at site f, upstream of the outfall, both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.16). The observed densities of ≥1+ salmon at site f both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.16). The observed densities of 0+ trout at site f both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CL was <1 for the pre-commissioning scenarios therefore the observed population was significantly lower than expected in this case (Table 4.16). The observed density of ≥1+ trout (<20 cm) at site f prior to commissioning of the hydropower scheme (2006-2007 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.16). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2009-2011 data), was higher than predicted (HQS), suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.16). The observed densities of ≥1+ trout (>20 cm) at site f both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were zero and lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.16).

The observed densities of 0+ salmon at site g, downstream of the outfall, both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.17). The observed densities of ≥1+ salmon at site g both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.17).

Table 4.16 HABSCORE outputs for site f (abstracted reach) before (2006-2007) and after (2009-2011) the Rottal Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2006-2007	35.0	11.44	13.71	3.93	47.76	0.83	0.11	6.25	-0.18
2009-2011	29.8	11.95	53.43	14.45	197.62	0.22	0.03	1.74	-1.51
≥1+ salmon									
2006-2007	92.0	30.07	9.05	2.69	30.46	3.32	0.53	20.85	1.19
2009-2011	68.4	27.48	8.43	2.48	28.62	3.26	0.52	20.48	1.18
0+ trout									
2006-2007	5.0	1.63	15.27	3.99	58.44	0.11	0.02	0.72	-2.20
2009-2011	16.58	6.64	10.36	4.19	63.97	0.41	0.06	2.77	-0.89
≥1+ trout (<20 cm)									
2006-2007	12.0	3.92	12.66	2.91	55.10	0.31	0.05	1.92	-1.11
2009-2011	22.2	8.91	5.99	1.38	26.01	1.49	0.24	9.07	0.40
≥1+ trout (>20 cm)									
2006-2007	0.0	0.00	0.36	0.11	1.16	0.91	0.28	2.91	-0.09
2009-2011	0	0	0.30	0.09	0.93	1.36	0.43	4.28	0.31

Table 4.17 HABSCORE outputs for site g (control) before (2006-2007) and after (2009-2011) the Rottal Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2006-2007	16.0	7.90	37.21	10.65	129.95	0.21	0.03	1.61	-1.56
2009-2011	37.9	16.35	122.56	28.96	518.69	0.13	0.02	1.14	-2.04
≥1+ salmon									
2006-2007	37.0	18.27	11.52	3.43	38.62	1.59	0.25	10.04	0.46
2009-2011	75.3	32.49	6.92	2.00	23.98	4.69	0.73	30.05	1.55
0+ trout									
2006-2007	0.0	0.00	16.83	4.46	63.41	0.03	0.00	0.19	-3.50
2009-2011	14.2	6.12	10.03	2.60	38.62	0.61	0.09	4.12	-0.50
≥1+ trout (<20 cm)									
2006-2007	12.0	5.93	9.13	2.12	39.41	0.65	0.10	4.03	-0.43
2009-2011	7.0	3.01	3.78	0.85	16.73	0.80	0.13	4.96	-0.22
≥1+ trout (>20 cm)									
2006-2007	0.0	0.00	0.28	0.09	0.87	1.79	0.57	5.63	0.58
2009-2011	0	0	0.31	0.10	0.97	1.41	0.44	4.50	0.34

The observed densities of 0+ trout at site g both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CL was <1 for the pre-commissioning scenarios therefore the observed population was significantly lower than expected in this case (Table 4.17). The observed densities of ≥1+ trout (<20 cm) at site g both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.17). The observed densities of ≥1+ trout (>20 cm) at site g both prior to and following commissioning of the hydropower scheme (2006-2007 and 2009-2011 data) were zero and lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.17).

4.4.4 Salmonid growth rates

It should be noted that the growth calculations at sites a and b for 3 and 4 year old fish were based on small numbers of fish, often only one individual, so were subject to wide variation

Upstream of the intake

At site a, upstream of the intake on Burn of Heughs, back-calculated length of trout at age 1 ranged between 48 and 68 mm, with the slowest growth in 2006 and fastest in 2008 and 2009 (Table 4.18). Back-calculated length of trout at age 2 ranged between 109 and 132 mm, with the slowest growth in 2007 and fastest in 2009 (Table 4.18).

At site c, upstream of the intake on Kennel Burn, back-calculated length of trout at age 1 ranged between 49 and 75 mm, with the slowest growth in 2007 and fastest in 2004, although the latter value was based on one individual (Table 4.18). Back-calculated length of trout at age 2 ranged between 102 and 140 mm, with the slowest growth in 2007 and fastest in 2008 (Table 4.18).

Depleted reach

At site b, downstream of the intake on Burn of Heughs, back-calculated length of trout at age 1 ranged between 52 and 67 mm; with the slowest growth in 2006 and fastest in 2010 (Table 4.18). Back-calculated length of trout at age 2 ranged between 101 and 120 mm, with the slowest growth in 2006 and fastest in 2008.

Table 4.18 Back-calculated lengths at age of trout in the upper and lower reaches of Rottal Burn. All sites at Rottal in 2007 are excluded in the ageing data set as sites were combined and therefore not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))			
		1	2	3	4
a	2005	60±11(17)			
	2006	48(1)	114(1)	150(1)	
	2007	51±8(2)	109±4(2)		
	2008	61±10(18)	114±12(14)		
	2009	68±7(22)	132±10(4)		
	2010	68±12(13)			
	*2011	61±6(3)			
b	2003	66(1)	112(1)	158(1)	
	2004	64±8(8)	113±10(8)		
	2005	57±11(16)			
	2006	52±9(2)	101±7(2)	134±18(2)	
	2007	54±9(10)	117±21(10)	210(1)	266(1)
	2008	62±8(16)	120±15(5)	196(1)	
	2009	65±10(23)	114±12(7)		
	2010	67±9(13)			
	*2011	54±5(10)			
c	2004	75(1)	132(1)		
	2005	62±10(15)			
	2006	57±0(2)	105±4(2)	155± 3(2)	
	2007	49±4(4)	102±5(4)		
	2008	66±5(5)	140±14(4)		
	2009	65±9(25)	121±10(11)		
	2010	61±4(4)			
	*2011	63±5(11)			
d	2003	68(1)	135(1)	190(1)	
	2004	68±8(2)	125±10(2)		
	2005	59±12(18)			
	2006	55(1)	123(1)	184(1)	
	2007	60±8(8)	118±11(8)		
	2008	64±9(3)	119±8(2)		
	2009	69±9(15)			
	2010	70±12(17)			
	*2011	59±5(19)			
e	2005	62±4(3)	107±10(3)		
	2006	59±10(13)	123(1)	172(1)	
	2007	65±10(7)	127±14(7)	190(1)	
	2008	66±7(14)			
	2009	73±12(14)	117(1)		
	2010	62±4(9)			
	*2011	55±9(4)			
f	2005	55(1)	98(1)	147(1)	184(1)
	2006	62(1)	99(1)	142(1)	
	2007	65±17(3)	120±13(3)		
	2008	60±10(12)	134(1)		
	2009	69±12(27)	110±12(2)		
	2010	67±7(9)			
	*2011	47±6(12)			
g	2004	49±7(2)	96±13(2)		
	2005	52±9(12)			
	2007	54(1)	108(1)		
	2008	69±13(12)	141(1)		
	2009	63±6(8)	123(1)		
	*2011	58±8(12)			

Back-calculated length of trout at age 3 ranged between 134 and 210 mm, with the slowest growth in 2006 and fastest in 2007 (Table 4.18).

At site d, downstream of the intake on Kennel Burn, back-calculated length of trout at age 1 ranged between 55 and 70 mm; with the slowest growth in 2006 and fastest in 2010 (Table 4.18). Back-calculated length of trout at age 2 ranged between 118 and 135 mm, with the slowest growth in 2007 and fastest in 2003 (Table 4.18). Back-calculated length of trout at age 3 ranged between 184 and 190 mm. It should be noted that the growth calculations at sites a and b for three and four year old fish were based on small numbers of fish, often only one individual, so were subject to wide variation.

At site e, in the abstracted reach, back-calculated length of trout at age 1 ranged between 55 and 73 mm with the slowest growth in 2006, excluding 2011 data as this was based on observed mean length, and fastest in 2009 (Table 4.18). Back-calculated length of trout at age 2 ranged between 107 and 127 mm, with the slowest growth in 2005 and fastest in 2007 (Table 4.18). Back-calculated length of trout at age 3 ranged between 172 and 190 mm, with the slowest growth in 2006 and fastest in 2007 (Table 4.18). It should be noted that the growth calculations for three year old fish were based on small numbers of fish, often only one individual, so were subject to wide variation. Back-calculated length of salmon at age 1 ranged between 35 and 68 with the slowest growth in 2005, albeit based on one fish, and fastest in 2009 (Table 4.19). Back-calculated length of salmon at age 2 ranged between 82 and 97 mm, with the slowest growth in 2005 and fastest in 2008.

At site f, back-calculated length of trout at age 1 ranged between 47 and 69 mm; with the slowest growth in 2005, excluding 2011 data as this was based on observed mean length, and fastest in 2009 (Table 4.18). Back-calculated length of trout at age 2 ranged between 98 and 134 mm, with the slowest growth in 2005 and fastest in 2008 (Table 4.18). Back-calculated length of trout at age 3 ranged between 142 and 147 mm, and at age 4 was 184 mm in 2005 (Table 4.18). Back-calculated length of salmon at age 1 ranged between 44 and 62 with the slowest growth in 2010 excluding 2011 data as this was based on observed mean length, and fastest in 2009 (Table 4.19); Back-calculated length of salmon at age 2 was 88 mm in 2009.

Downstream of the outfall

At site g, back-calculated length of trout at age 1 ranged between 49 and 69 mm; with the slowest growth in 2004 and fastest in 2008 (Table 4.18). Back-calculated length of trout at age 2 ranged between 96 and 141 mm, with the slowest growth in 2007 and fastest in 2008 (Table 4.18). Back-calculated length of salmon at age 1 ranged between 45 and 62 mm with the slowest growth in 2007 and fastest in 2009 (Table 4.19). Back-calculated length of salmon at age 2 ranged between 91 and 109 mm with the slowest growth in 2008 and fastest in 2009 (Table 4.19).

Overall during the study period the growth of trout and salmon within the abstracted reaches followed similar trends to the growth of trout and salmon at sites outside the abstracted reach.

Table 4.19 Back-calculated lengths at age of salmon at sites e, f and g in the lower reaches of Rottal Burn. All sites at Rottal in 2007 are excluded in the ageing data set as sites were combined and therefore not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))		
		1	2	3
e	2005	35(1)	82(1)	
	2006	49±6(11)		
	2007	44±9(6)	89±9(6)	119(1)
	2008	56±3(9)	97(1)	
	2009	68±6(4)		
	2010	63±6(5)		
	*2011	50±4(5)		
f	2009	62±9(28)	88±3(2)	
	2010	51±8(12)		
	*2011	44±2(8)		
g	2005	46±10(15)		
	2007	45±7(4)	94±6(4)	
	2008	58±9(14)	91±1(2)	
	2009	62±6(27)	109(1)	
	2010	59±11(15)		
	*2011	47±5(25)		

4.4.5 Eel

Eels were only captured in the lower reaches associated with the outfall on Rottal Burn. Numbers of eels captured at sites f and g varied between survey years and data suggest the lower reaches of Rottal Burn are used by small numbers of eels (Table 4.20; Figure 4.28). At site f and g numbers of eels caught over the study period were similar pre and post hydropower commissioning. Despite some variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers with the hydropower scheme operational.

Table 4.20 Number of eel captured at survey sites in the lower reaches of Rottal Burn between 2006 and 2011.

Year	Site identifier	
	f	g
2006	12	4
2007	15	2
2009	10	0
2010	4	3
2011	2	1

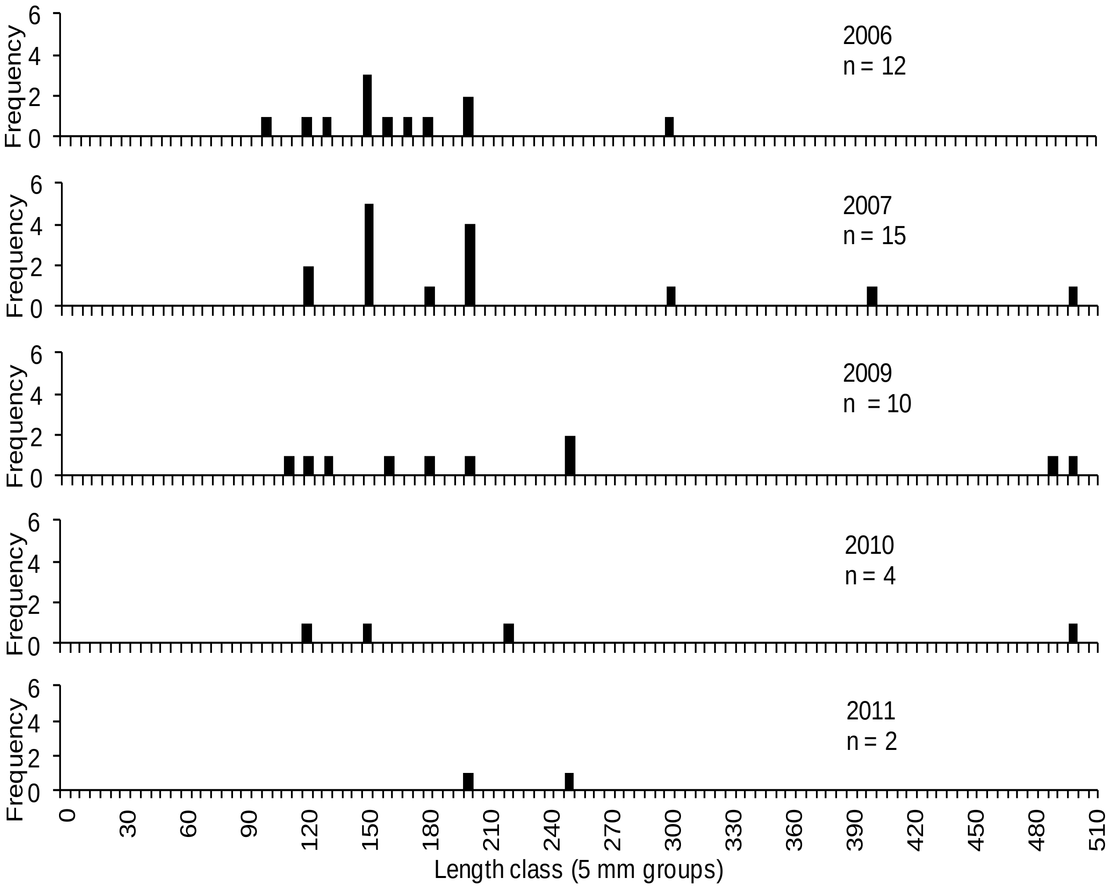

Figure 4.28 Length distributions of eel at site f in Rottal Burn between 2006 and 2011. No surveys were carried out in 2008.

4.4.6 Wetted widths

The wetted widths from site f, as this was located within the abstracted reach, were statistically tested for any differences between HABSCORE surveys pre and post commissioning. As data did not conform to normal distribution or homogeneity of variance a Mann-Whitney *U*-test was performed. Confidence limits enclosed zero (-0.100, 1.600) with a p value of 0.9, n = 26, therefore no significant difference was detected between the wetted widths collected during HABSCORE surveys before and after commissioning of the hydropower scheme, confirming comparability of density estimates over the study period.

4.5 Innerhadden Burn hydropower scheme

4.5.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

No salmon were captured in the upper reaches associated with the intakes on Innerhadden Burn throughout the study period.

Upstream of the intakes

At site a, upstream of the intake on Glen Sassunn Burn, 0+ trout were absent (class F) throughout the study period with the exception of 2005 when densities were poor (class E) (Figure 4.29, Table 4.21). ≥1+ trout densities varied over the study period with the greatest density in 2007 and the lowest in 2010; populations were fair/poor (class D) in 2006, poor (class E) in 2005 and 2010 and absent (class F) in other years (Figure 4.29, Table 4.21).

0+ trout densities at site c, upstream of the intake on Allt Coire Cruach Sneachda, were low or zero during the study period and classified as poor (class E) in 2005 and 2007 and absent (class F) in other years (Figure 4.29, Table 4.21). ≥1+ trout were absent in 2005 and 2006 and found in low densities in other years, varying from fair/poor (class D) to poor (class E) in other years.

Depleted reach

At site b, downstream of the intake on Glen Sassunn Burn, 0+ trout densities were higher than at site a in all years (Figure 4.29). 0+ trout populations were poor (class E) in 2005 improving to fair/poor (class D) in 2006 before declining to zero in 2007; 0+ trout populations were poor (class E) in 2010 and zero in 2011 (Table 4.21). ≥1+ trout densities at site b varied over the study period with the greatest density in 2007 and the lowest in 2011 however populations were higher or comparable to those found at site a (Figure 4.29, Table 4.21). ≥1+ trout populations were fair/average (class C) between 2005 and 2010 but fair/poor (class D) in 2011 (Table 4.21).

At site d, downstream of the intake on Allt Coire Cruach Sneachda, 0+ trout densities were higher than at site c but overall showed a decrease during the study period (Figure 4.29). 0+ trout populations were fair/poor (class D) between 2005 and 2007 decreasing to poor (class E) in 2010 and zero in 2011 (Table 4.21); this trend reflected the findings at site c upstream of the intake. ≥1+ trout densities varied over the study period with the greatest density in 2007 and the lowest in 2010 (Figure 4.29). ≥1+ trout populations were excellent (class A) in 2007 fair/average (class C) in 2005, 2006 and 2011 and fair/poor (class D) in 2010 (Table 4.21).

Table 4.21 Classification of trout densities in both the upper and lower reaches of Innerhadden Burn between 2005 and 2011 based on five classification schemes. Note hydropower scheme commissioned November 2009. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2005	E	E	E	E	E
	2006	F	F	F	F	F
	2007	F	F	F	F	F
	2008	n/s	n/s	n/s	n/s	n/s
	2010	F	F	F	F	F
	2011	F	F	F	F	F
b*	2005	E	E	D	E	D
	2006	D	E	D	E	D
	2007	F	F	F	F	E
	2008	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	F	F	F	F	F
c	2005	E	E	E	E	E
	2006	F	F	F	F	F
	2007	E	E	E	E	E
	2008	n/s	n/s	n/s	n/s	n/s
	2010	F	F	F	F	F
	2011	F	F	F	F	F
d*	2005	D	E	D	E	D
	2006	D	E	E	E	D
	2007	D	E	E	E	D
	2008	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	F	F	F	F	F
e *	2005	E	E	E	E	E
	2006	E	E	E	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2010	E	E	E	E	E
	2011	E	E	E	E	E
f	2005	F	F	F	F	F
	2006	E	E	E	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	D
	2010	F	F	F	F	F
	2011	E	E	E	E	E

≥1+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2005	C	C	D	D	C
	2006	D	D	D	E	D
	2007	B	C	C	D	C
	2008	n/s	n/s	n/s	n/s	n/s
	2010	F	F	F	F	F
	2011	D	D	E	E	C
b*	2005	C	C	B	B	C
	2006	C	C	B	B	C
	2007	C	C	B	B	C
	2008	n/s	n/s	n/s	n/s	n/s
	2010	C	D	E	E	C
	2011	D	D	C	D	D
c	2005	F	F	F	F	F
	2006	F	F	F	F	F
	2007	D	D	E	E	D
	2008	n/s	n/s	n/s	n/s	n/s
	2010	E	E	E	E	E
	2011	E	E	E	E	E
d*	2005	C	C	B	B	C
	2006	C	C	C	D	C
	2007	A	B	B	C	B
	2008	n/s	n/s	n/s	n/s	n/s
	2010	D	D	E	E	D
	2011	C	C	E	E	D
e*	2005	F	F	F	F	F
	2006	E	E	C	C	E
	2007	E	E	C	C	E
	2008	E	E	E	D	E
	2010	D	E	C	B	D
	2011	E	E	C	C	E
f	2005	F	F	F	F	F
	2006	E	E	E	E	E
	2007	E	E	D	C	E
	2008	F	F	F	F	F
	2010	E	E	E	E	E
	2011	E	E	E	E	E

Salmonid populations in the lower reaches of Innerhadden Burn (sites e and f) were dominated by salmon with lower numbers of trout.

At site e, upstream of the outfall, 0+ salmon populations were highest in 2010 and lowest in 2005 ranging from fair/average (class C) in 2010 to poor (class E) in 2005, 2006 and 2011; 0+ salmon populations in 2010 and 2011 were higher than in 2005 and 2006 (Figure 4.29, Table 4.22). ≥1+ salmon populations were highest in 2007 and lowest in 2005, with populations in 2010 and 2011 higher than in 2005, 2006 and 2008 (Figure 4.29). ≥1+ salmon populations ranged between poor (class E) in 2005 to excellent (class A) in 2007 and 2011 (Table 4.22). 0+ trout populations were lower than salmon and generally stable but were highest in 2010 and lowest in 2005 and classified as poor (class E) in all years (Figure 4.29,Table 4.21); there was no variation in the classifications whichever scheme was used. ≥1+ trout populations were also lower than salmon during the study period with the highest in 2010 and zero in 2005 ranging from absent (class F) in 2005 to poor (class E) between 2006 and 2008 and in 2011; ≥1+ trout populations were fair/poor (class D) in 2010 (Figure 4.29,Table 4.21).

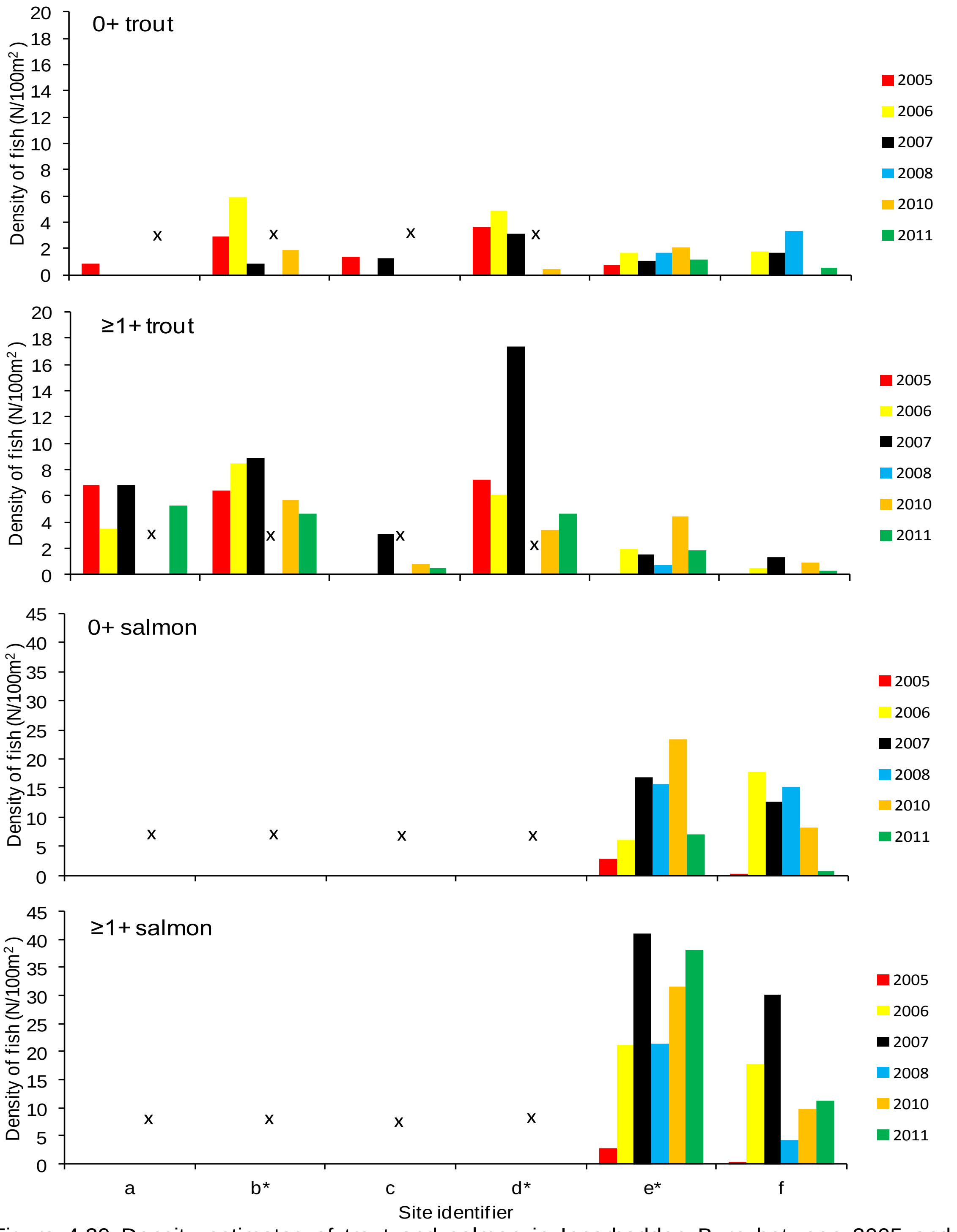

Figure 4.29 Density estimates of trout and salmon in Innerhadden Burn between 2005 and 2011. Note sites a-d were not sampled in 2008 and no sites were sampled in 2009. Hydropower scheme was commissioned November 2009. x represents years not sampled.

Downstream of the outfall

At site f, 0+ salmon populations were highest in 2006 and lowest in 2005 with a similar trend of low 0+ salmon densities in 2011 as found at site e (Figure 4.29). 0+ salmon populations were poor (class E) in 2005 and 2011 and fair/poor (class D) in other years (Table 4.22). ≥1+ salmon populations were highest in 2007 and lowest in 2005 with a similar trend to those found at site e (Figure 4.29). ≥1+ salmon populations were excellent (class A) in 2007, fair/average (class C) in 2006 and 2011, but either fair/average (class D) or poor (class E) in other years (Table 4.22).

At site f, 0+ trout populations were low throughout the study period as found at site e with an absence of 0+ trout in 2005 and 2010 (Figure 4.29). 0+ trout populations were poor (class E) in all years except 2005 and 2010 when no 0+ trout were captured. ≥1+ trout populations were low or zero during the study period with a similar trend to those found at site e (Figure 4.29). ≥1+ trout populations were poor (class E) in all years except 2005 and 2008 when no ≥1+ trout were captured (Table 4.21).

Table 4.22 Classification of salmon densities in the lower reaches of Innerhadden Burn between 2005 and 2011, based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
e*	2005	E	E	E	E	E
	2006	E	E	E	E	E
	2007	D	E	D	D	D
	2008	D	E	D	E	D
	2010	C	D	C	D	C
	2011	E	E	E	E	E
f	2005	E	E	E	E	E
	2006	D	D	D	D	D
	2007	D	E	D	D	D
	2008	D	E	D	D	D
	2010	D	E	D	E	E
	2011	E	E	E	E	E

≥1+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
e*	2005	E	E	E	E	D
	2006	B	C	B	C	A
	2007	A	A	A	A	A
	2008	D	D	C	D	B
	2010	B	B	B	B	A
	2011	A	B	A	A	A
f	2005	E	E	E	E	E
	2006	C	D	C	C	B
	2007	A	A	A	A	A
	2008	D	E	D	D	D
	2010	D	D	D	D	C
	2011	C	D	D	D	B

4.5.2 Salmonid size structure

Upstream of the intakes

Length distributions of trout in the upper reaches associated with the two intakes varied between years at each site. At site a, upstream of the intake on Glen Sassunn Burn, only one 0+ trout was caught during the study period while ≥1+ trout were caught in the size range 85-160 mm between 2005 and 2007 and a narrower size range of 95-120 mm in 2011. At site c, upstream of the intake on Allt Coire Cruach Sneachda, 0+ and ≥1+ trout were caught in too few numbers for clear discrimination of size structure.

Depleted reach

At site b, downstream of the intake on Glen Sassunn Burn, 0+ trout were caught in the size range 45-60 mm and 45-65 mm in 2005 and 2006 respectively with a narrower size range in other years due to the low numbers present (Figure 4.30). ≥1+ trout were caught in a similar size range (80-160 mm) throughout the study period except in 2010 when a small number of larger individuals >160 mm were captured (Figure 4.30).

At site d, downstream of the intake on Allt Coire Cruach Sneachda, 0+ trout were caught in the size range 45-65 mm in 2005 and 2006 and in a narrower size range in 2007 and 2010 (Figure 4.31). ≥1+ trout were caught in a similar size range (80-160 mm) in 2005, 2006 and 2007 and a narrower size range in 2010 and 2011 (Figure 4.31).

Length distributions of salmon in the lower reaches of Innerhadden Burn were similar between years, albeit in varying numbers, at both sites e and f (Figures 4.32 & 4.33).

At site e, upstream of the outfall, 0+ salmon were caught in a narrow size range of 50-70 mm in 2005 and 2006 and a wider size range of 40-70 mm in subsequent years; ≥1+ salmon were caught in the size range 80-130 mm, throughout the study period except in 2005 when ≥1+ salmon were in a narrower size range of 85-100 mm (Figure 4.32). 0+ trout were caught in a similar size range of 40-70 mm between 2006 and 2008 and in 2010, with a slightly narrower size range in other years (Figure 4.34). ≥1+ trout were caught in a wide size range (80-160 mm) in 2007, 2010 and 2011 but a narrow size range in other years (Figure 4.34).

Downstream of the outfall

At site f, downstream of the outfall, 0+ salmon were caught in a narrow size range of 35-55 mm in 2011 and a wider size range of 35-65 mm in subsequent years except in 2005 when only one 0+ salmon was caught (Figure 4.33). ≥1+ salmon were caught in the size range 75-125 mm, throughout the study period except in 2005 when only one ≥1+ salmon was caught (Figure 4.33). 0+ trout were absent in 2010 and caught in variable size ranges in other years; ≥1+ trout were also caught in variable numbers with no clear trends in size structure (Figure 4.35).

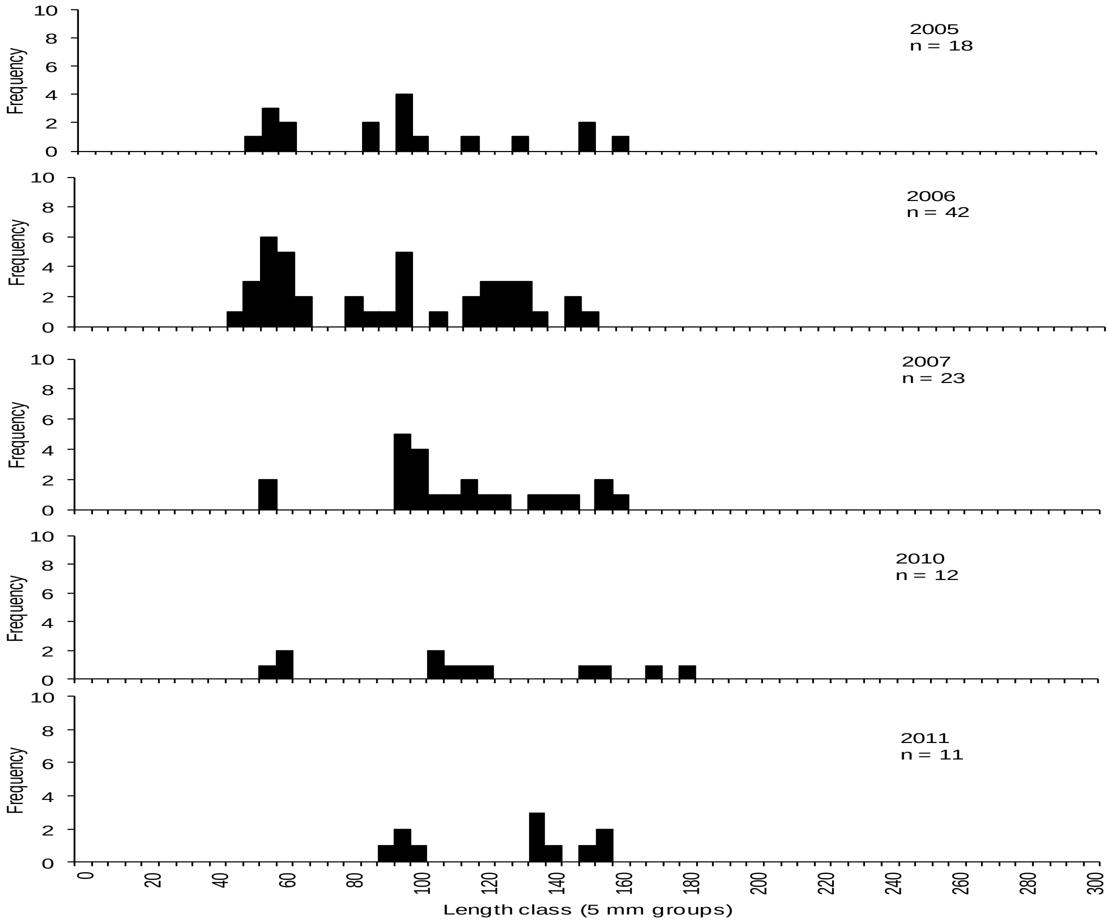

Figure 4.30 Length distributions of trout at site b in Innerhadden Burn between 2005 and 2011. Note site b was not surveyed in 2008 and 2009.

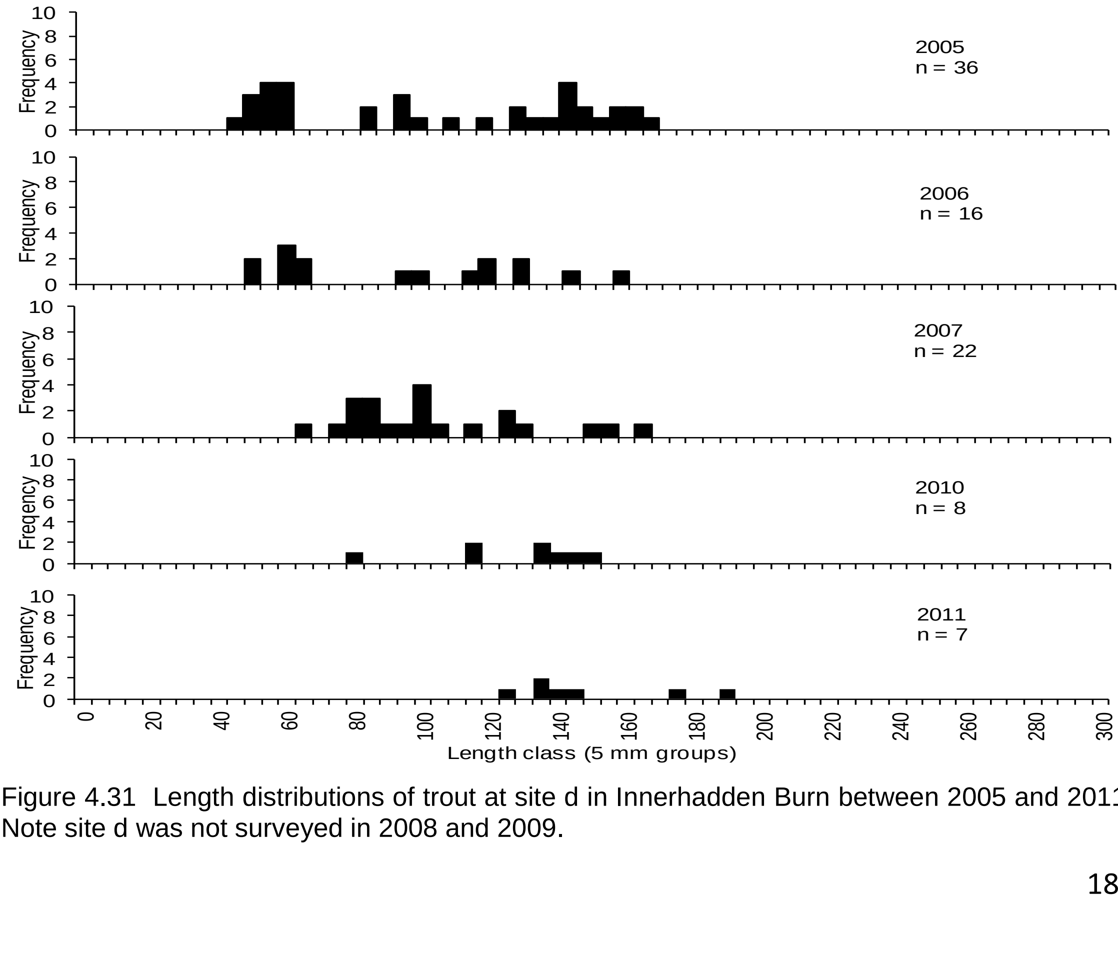

Figure 4.31 Length distributions of trout at site d in Innerhadden Burn between 2005 and 2011. Note site d was not surveyed in 2008 and 2009.

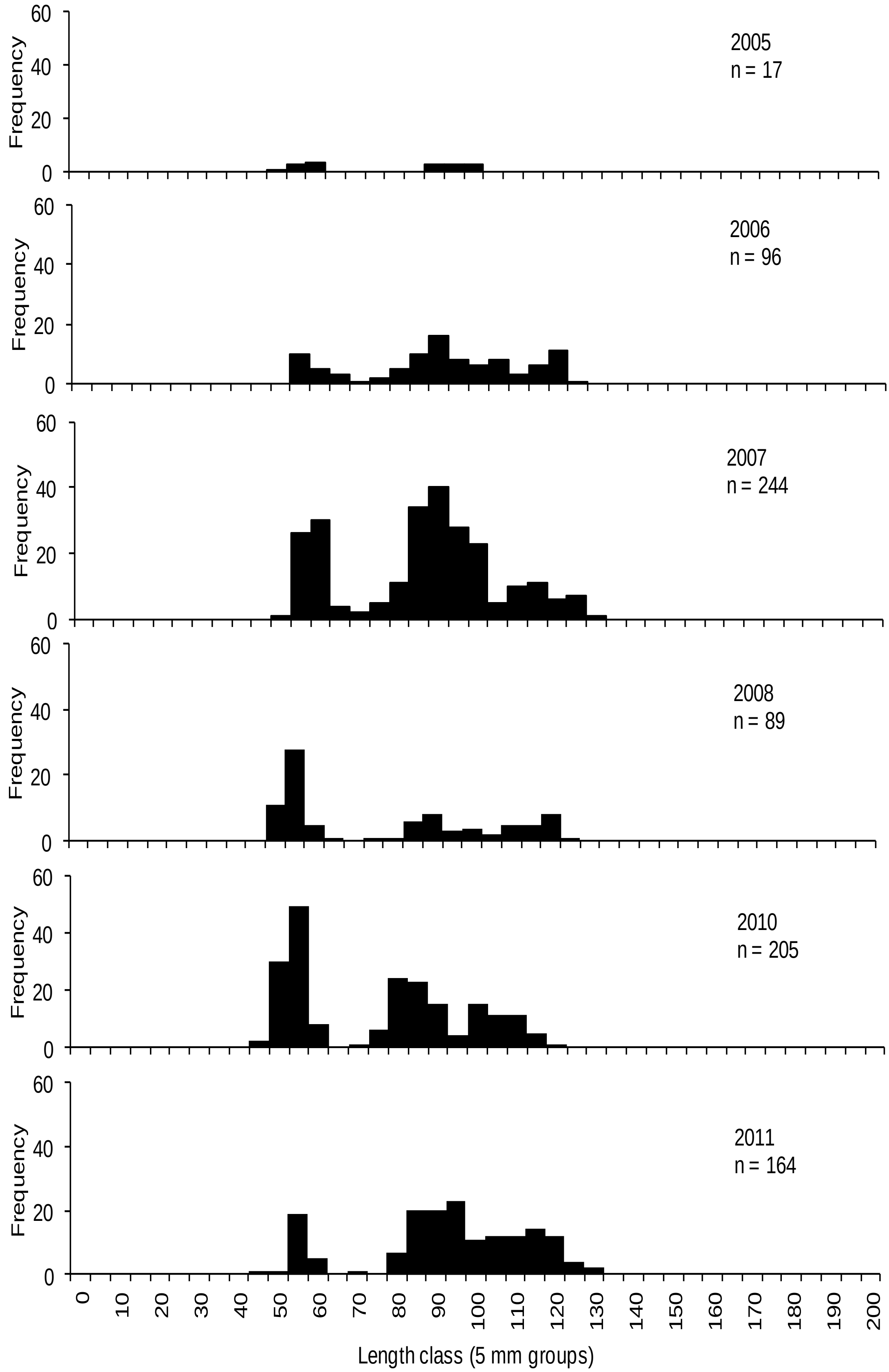

Figure 4.32 Length distributions of salmon at site e in Innerhadden Burn between 2005 and 2011. Note site e was not surveyed in 2009.

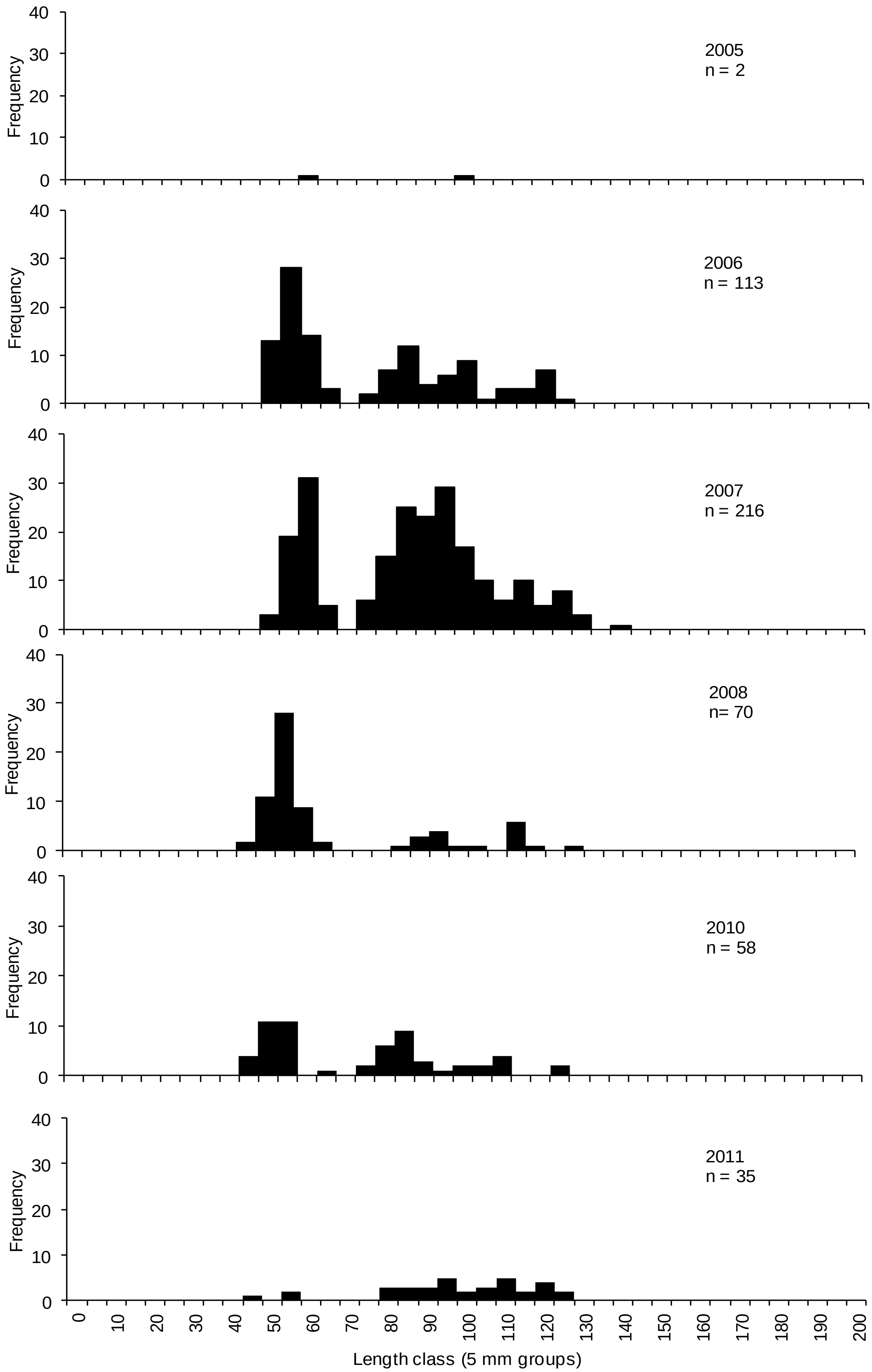

Figure 4.33 Length distributions of salmon at site f in Innerhadden Burn between 2005 and 2011. Note site f was not surveyed in 2009.

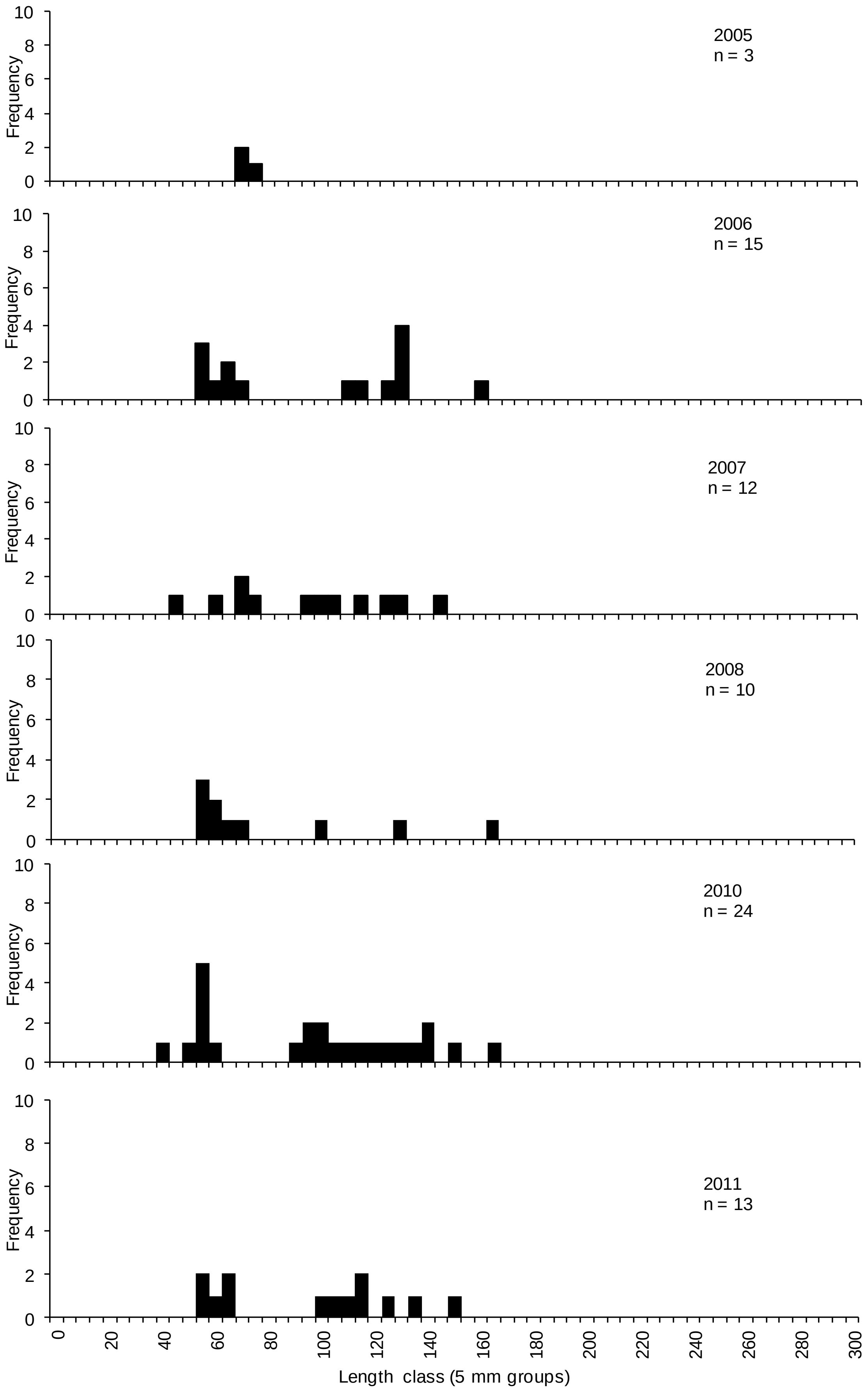

Figure 4.34 Length distributions of trout at site e in Innerhadden Burn between 2005 and 2011. Note site e was not surveyed in 2009.

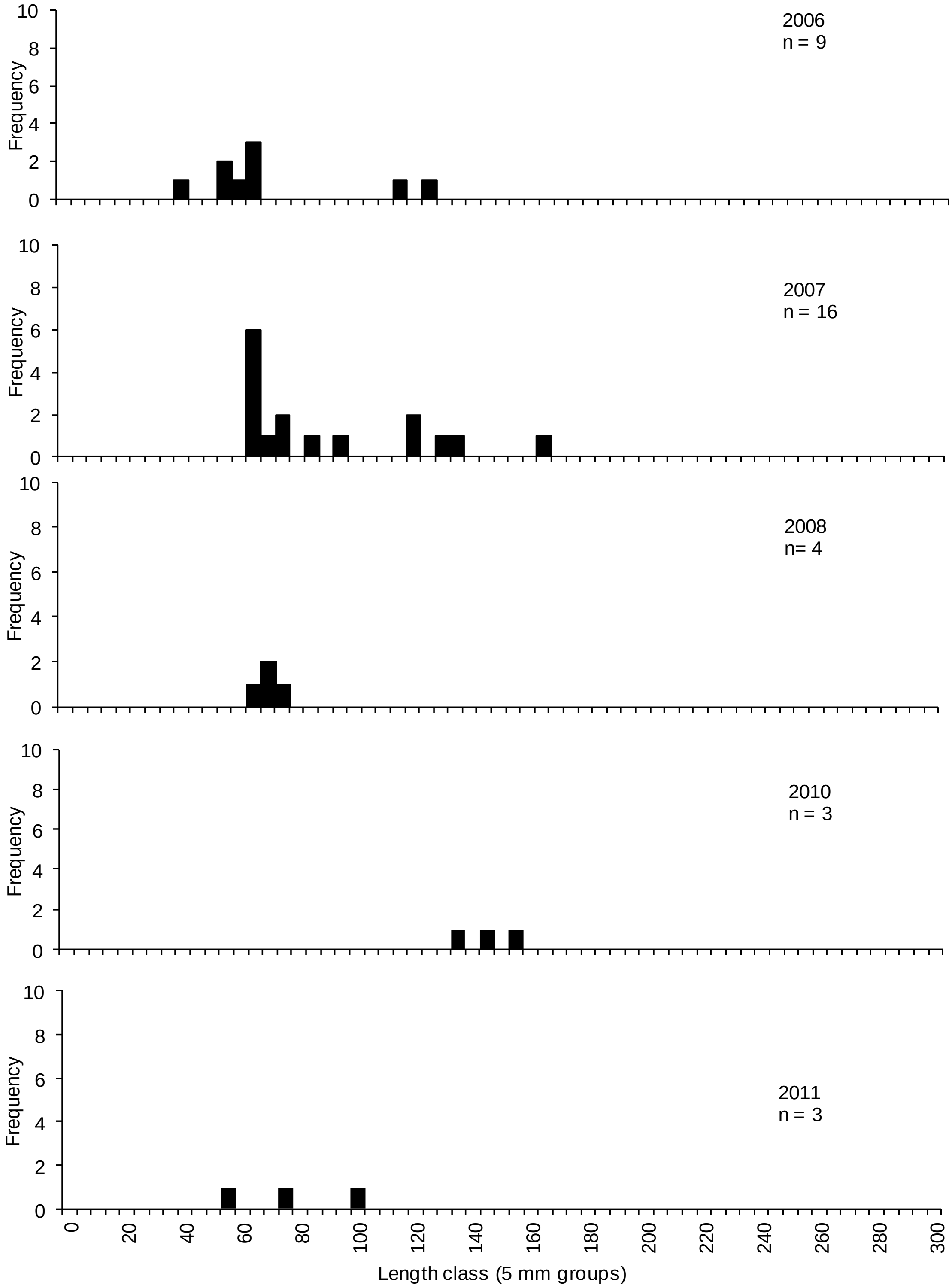

Figure 4.35 Length distributions of trout at site f in Innerhadden Burn between 2005 and 2011. Note site f was not surveyed in 2009.

4.5.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was commissioned were compared to the same information collected following commissioning at two key sites in the lower reaches of Innerhadden Burn. For the analysis habitat data collected in 2008 were utilised with fisheries data collected from

2005-2008 and compared to habitat data collected in 2011 utilised with fisheries data collected from 2010 and 2011 (Tables 4.23 & 4.24). The data allowed comparison between one site in the abstracted reach and one site downstream of the abstracted reach.

The observed densities of 0+ salmon at site e, upstream of the outfall, both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.23). The observed densities of ≥1+ salmon at site e both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.23). The observed densities of 0+ trout at site e both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CL was <1 for both scenarios therefore the observed populations were significantly lower than expected (Table 4.23). The observed density of ≥1+ trout (<20 cm) at site e prior to commissioning of the hydropower scheme (2005-2008 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.23). The observed density of ≥1+ trout (<20 cm) following commissioning of the hydropower scheme (2010-2011 data), was higher than predicted (HQS), suggesting better populations than expected, but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.23). The observed densities of ≥1+ trout (>20 cm) at site e both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were zero and lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.23).

The observed densities of 0+ salmon at site f, downstream of the outfall, both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CL was <1 in the latter scenario therefore the observed population was significantly lower than expected (Table 4.24).

Table 4.23 HABSCORE outputs for site e (abstracted reach) before (2005-2008) and after (2010-2011) the Innerhadden Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2008	46.5	11.52	28.87	7.75	107.56	0.40	0.05	3.12	-0.91
2010-2011	53.9	13.46	56.59	14.80	216.35	0.24	0.03	1.89	-1.43
≥1+ salmon									
2005-2008	63.2	15.65	5.79	1.66	20.27	2.70	0.42	17.36	0.99
2010-2011	147.5	36.88	6.27	1.80	21.80	5.88	0.92	37.60	1.77
0+ trout									
2005-2008	5.9	1.46	10.52	2.63	42.06	0.14	0.02	0.96	-1.96
2010-2011	6.3	1.58	11.37	2.88	44.48	0.14	0.02	0.96	-1.97
≥1+ trout (<20 cm)									
2005-2008	2.7	0.68	2.62	0.60	11.45	0.26	0.04	1.58	-1.34
2010-2011	12.0	3.00	1.85	0.42	8.13	1.63	0.76	10.04	0.49
≥1+ trout (>20 cm)									
2005-2008	0.0	0.00	0.21	0.07	0.66	1.18	0.37	3.75	0.16
2010-2011	0	0	0.26	0.08	0.81	0.96	0.31	2.97	-0.04

Table 4.24 HABSCORE outputs for site f (control) before (2005-2008) and after (2010-2011) the Innerhadden Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2005-2008	64.9	15.91	23.56	6.42	86.38	0.68	0.09	5.24	-0.34
2010-2011	9.2	2.79	23.60	6.65	83.77	0.12	0.02	0.90	-2.12
≥1+ salmon									
2005-2008	54.3	13.32	6.34	1.82	22.08	2.10	0.33	13.41	0.74
2010-2011	35.9	10.93	6.83	1.97	23.65	1.60	0.25	10.24	0.47
0+ trout									
2005-2008	6.7	1.64	9.77	2.47	38.57	0.17	0.02	1.17	-1.77
2010-2011	1.4	0.43	16.28	4.11	64.54	0.03	0.00	0.18	-3.51
≥1+ trout (<20 cm)									
2005-2008	2.6	0.65	3.93	0.90	17.20	0.17	0.03	1.01	-1.77
2010-2011	1.7	0.53	4.84	1.11	21.05	0.11	0.02	0.66	-2.21
≥1+ trout (>20 cm)									
2005-2008	0.0	0.00	0.20	0.06	0.65	1.21	0.38	3.87	0.19
2010-2011	0	0	0.27	0.09	0.87	1.12	0.35	3.55	0.11

The observed densities of ≥1+ salmon at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.24). The observed densities of 0+ trout and ≥1+ trout (<20 cm) at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected and the HUI upper CLs were <1 for the post-commissioning scenarios for both age groups therefore the observed populations were significantly lower than expected (Table 4.24). The observed densities of ≥1+ trout (>20 cm) at site f both prior to and following commissioning of the hydropower scheme (2005-2008 and 2010-2011 data) were zero and lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CLs were >1 in both scenarios therefore the observed populations were not significantly lower than expected (Table 4.24).

4.5.4 Salmonid growth rates

It should be noted that the growth calculations at sites a and b for three and four year old fish were based on small numbers of fish, often only one individual, so were subject to wide variation.

Upstream of the intakes

At site a back-calculated length of trout at age 1 ranged between 51 and 65 mm, with the slowest growth in 2002 and fastest in 2010 (Table 4.25). Back-calculated length of trout at age 2 ranged between 98 and 110 mm, with the slowest growth in 2003 and fastest in 2002; back-calculated length of trout at age 3 was 150 mm in 2002 (Table 4.25). At site c back-calculated length of trout at age 1 ranged from between 67 and 69 mm, and back-calculated length of trout at age 2 was 133 mm (Table 4.25). It should be noted that the growth calculations at site c were based on fish in each case.

Depleted reach

At site b back-calculated length of trout at age 1 ranged between 53 and 69 mm, with the slowest growth in 2004 and fastest in 2005 and 2010 (Table 4.25). Back-calculated length of trout at age 2 ranged between 96 and 119 mm, with the slowest growth in 2003 and 2004 and fastest in 2008; back-calculated length of trout at age 3 ranged between 122 and 149 mm (Table 4.25). At site d back-calculated length of trout at age 1 ranged between 45 and 76 mm with the slowest growth in 2001 and fastest in 2010

(Table 4.25). Back-calculated length of trout at age 2 ranged between 98 and 111 mm, with the slowest growth in 2001 and fastest in 2002 (Table 4.25). Back-calculated length of trout aged 3 ranged between 134 and 149 mm, with trout at age 4 being 170 mm (Table 4.25).

Table 4.25 Back calculated lengths at age of trout in the upper and lower reaches of Innerhadden Burn. No 0+ trout were caught in 2011 at sites a-d. Ageing data from all sites in 2007 excluded as sites were combined and thus not comparable. Data from sites e and f in 2008 excluded as sites were combined and not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))			
		1	2	3	4
a	2002	51±4(5)	110± 14(5)	150±12(5)	
	2003	52(1)	98(1)		
	2004	58± 6(4)	109±4(3)		
	2005	60± 15(2)			
	2010	65± 2(4)			
b	2002	60± 11(2)	109±8(2)	149±1(2)	
	2003	59± 12(9)	96±15(9)	122±2(2)	
	2004	53± 4(8)	96±8(4)		
	2005	69± 11(3)			
	2008	59 (1)	119(1)		
	2009	62± 12(7)	112±1(2)		
	2010	69± 13(6)			
c	2004	69(1)			
	2005	67(1)			
	2009	67(1)	133 (1)		
d	2001	45(1)	98(1)	134(1)	170(1)
	2002	54±6 (8)	111±9(8)	149±11(8)	
	2003	49±10(7)	110±16(7)		
	2004	62±14(9)	105±20(3)		
	2005	67±10(4)			
	2009	71± 15(5)	102 ± 10 (2)		
	2010	76± 1(2)			
e	2003	59(1)	100(1)	142(1)	
	2004	63±7(7)	101±7(4)		
	2005	62±2(2)			
	2008	60(1)	119(1)		
	2009	67±12(16)	109±8 (3)		
	2010	72±4(5)			
	*2011	58±6(5)			
f	2004	48(1)	96(1)		
	2005	57(1)			
	2008	59±3(2)	113±2(2)		
	2009	79(1)			
	2010	65(1)			
	*2011	63±12(2)			

At site e back-calculated length of trout at age 1 ranged between 59 and 72 mm, with the slowest growth in 2003 and fastest in 2010 (Table 4.25). Back-calculated length of trout at age 2 ranged between 100 and 119 mm, with the slowest growth in 2003 and fastest in 2008; back-calculated length of trout at age 3 was 142 mm in 2003 (Table 4.25). Back-calculated length of salmon at age 1 ranged between 54 and 64 mm, with the slowest growth in 2003 and fastest in 2005 (Table 4.26). Back-calculated length of salmon at age 2 ranged between 87 and 98 mm, with the slowest growth in 2004 and fastest in 2009 (Table 4.26).

Downstream of the outfall

At site f back-calculated length of trout at age 1 ranged between 48 and 79 mm, with the slowest growth in 2004 and fastest in 2009; back-calculated length of trout at age 2 ranged between 96 and 113 mm (Table 4.25). Back-calculated length of salmon at age 1 ranged between 47 and 62 mm, with the slowest growth in 2004 and fastest in 2005; back-calculated length of salmon at age 2 ranged between 82 and 98 mm (Table 4.26).

Table 4.26 Back calculated lengths at age of salmon in the lower reaches of Innerhadden Burn. Ageing data from 2007 and 2008 excluded as sites e and f were combined and thus not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))	
		1	2
e	2003	54± 9(6)	91±5(6)
	2004	55±3(6)	87±3(2)
	2005	64±12(9)	
	2009	61±7(25)	98±3(3)
	2010	62±9(16)	
	*2011	54±4(27)	
f	2003	57(1)	98(1)
	2004	47±8(5)	82±8(4)
	2005	56±12(4)	
	2009	62±7(21)	
	*2011	50±4(3)	

4.5.5 Eel

No eel were caught in the upper or lower reaches of Innerhadden Burn throughout the study period.

4.5.6 Wetted widths

The wetted widths at site e, as this was located within the abstracted reach, were statistically tested for any differences collected during HABSCORE surveys pre and post commissioning. As data did not conform to normal distribution or homogeneity of variance a Mann-Whitney *U* test was performed. Confidence limits enclosed zero (-1.999, 2.499) with a p value of 0.3, n = 32. Therefore no significant difference was detected between the wetted widths collected during the surveys before and after commissioning of the hydropower scheme, confirming comparability of density estimates over the study period.

4.6 Inverhaggernie Burn hydropower scheme

4.6.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

No salmon were caught in the upper reaches of Inverhaggernie Burn throughout the study period.

Upstream of the intake

0+ trout densities at site a, upstream of the intake, were low or zero during the study period and classified as poor (class E) in 2006 and 2008 and absent (class F) in 2010 and 2011 (Figure 4.36, Table 4.27). ≥1+ trout densities at site a were also low throughout the study period ranging from fair/average (class C) in 2008 to poor (class E) in 2010 and 2011.

Depleted reach

0+ and ≥1+ trout densities were also low at site b, downstream of the intake, throughout the study period (Figure 4.36). 0+ trout populations were highest in 2008 and lower in 2010 and 2011; 0+ trout populations were poor (class E) except in 2008 when populations were fair/average (class C) (Table 4.27). The data suggest a decrease in 0+ trout populations since 2008; however these trends mirrored those found at site a upstream of the hydropower scheme. ≥1+ trout populations at site b followed a similar trend to those at site a, with the highest density in 2008 and lower densities in 2011; ≥1+ trout were absent in 2010 (Figure 4.36). ≥1+ trout populations varied from good (class B) in 2008 to absent (class F) in 2010 and poor (class E) in 2011 (Table 4.27).

Catches of salmonids in the lower reaches of Inverhaggernie Burn were dominated by salmon with lower densities of trout (Figure 4.36).

At site c, upstream of the outfall, 0+ salmon densities between 2006 and 2010 were generally stable and fair/average (class C), but decreased in 2011 to poor (class E) (Figure 4.36, Table 4.28); this trend of poor 0+ salmon was not found at site d downstream of the outfall. ≥1+ salmon populations were highest in 2006, generally stable between 2007 and 2010 and lower in 2011 (Figure 4.36); populations showed an overall decrease from excellent (class A) in 2006 to good (class B) in 2007 and 2008 and fair/average (class C) in 2010 and 2011 (Table 4.28). However, unlike for 0+

salmon, ≥1+ salmon populations at site c showed a similar trend to those at site d, downstream of the outfall, with populations at the latter site decreasing from good (class B) to fair/average (class C) (Table 4.28). At site c 0+ trout populations were lower than salmon and generally stable between 2007 and 2011, with the highest densities in 2006 and lowest in 2011; populations were fair/average (class C) in 2006, fair/poor (class D) between 2007 and 2010 and poor (class E) in 2011 and this decrease reflected findings at site d, downstream of the outfall (Figure 4.36, Table 4.27). ≥1+ trout populations at site c varied during the study period with the highest densities in 2011 and lowest in 2008 and 2011; populations ranged between good (class B) in 2007 and 2011 and poor (class E) in 2010 (Figure 4.36, Table 4.27).

Table 4.27 Classification of trout densities in both the upper and lower reaches of Inverhaggernie Burn between 2006 and 2011 based on five classification schemes. Note hydropower scheme commissioned in 2009. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2006	E	E	E	E	E
	2007	n/s	n/s	n/s	n/s	n/s
	2008	E	E	E	E	E
	2010	F	F	F	F	F
	2011	F	F	F	F	F
b*	2006	E	E	E	E	D
	2007	n/s	n/s	n/s	n/s	n/s
	2008	C	E	D	E	D
	2010	E	E	E	E	E
	2011	E	E	E	E	E
c*	2006	C	D	B	C	C
	2007	D	E	E	E	D
	2008	D	E	D	E	D
	2010	D	E	C	D	D
	2011	E	E	E	E	E
d	2006	n/s	n/s	n/s	n/s	n/s
	2007	D	D	C	D	D
	2008	D	E	D	E	D
	2010	E	E	E	E	E
	2011	D	E	D	E	D

≥1+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2006	D	D	D	E	D
	2007	n/s	n/s	n/s	n/s	n/s
	2008	C	C	C	D	D
	2010	E	E	E	E	E
	2011	E	E	E	E	E
b*	2006	D	E	D	E	D
	2007	n/s	n/s	n/s	n/s	n/s
	2008	B	C	B	C	C
	2010	F	F	F	F	F
	2011	E	E	E	E	D
c*	2006	C	C	B	B	C
	2007	B	C	B	C	C
	2008	C	C	C	D	D
	2010	E	E	D	E	D
	2011	B	B	A	A	B
d	2006	n/s	n/s	n/s	n/s	n/s
	2007	C	C	B	B	C
	2008	C	C	C	D	C
	2010	E	E	E	E	E
	2011	E	E	E	E	E

Downstream of the outfall

At site d, downstream of the outfall, 0+ salmon densities increased between 2007 and 2010 and densities in 2011 were similar to 2010; populations were fair/poor (class D) in 2007 and fair/average (class C) in other years (Figure 4.36, Table 4.28). ≥1+ salmon populations were similar in 2006, 2007 and 2011 but low in 2010 overall decreasing from good (class B) to fair/average (class C) (Figure 4.36,Table 4.28). 0+ trout populations showed an overall decrease during the study period as found at site c, and were classified as fair/poor (class D) in 2007, 2008 and 2011 and poor (class E) in 2010. ≥1+ trout populations showed an overall decrease during the study period and were classified as fair/average (class C) in 2007 and 2008 and poor (class E) in 2010 and 2011 (Figure 4.36, Table 4.27).

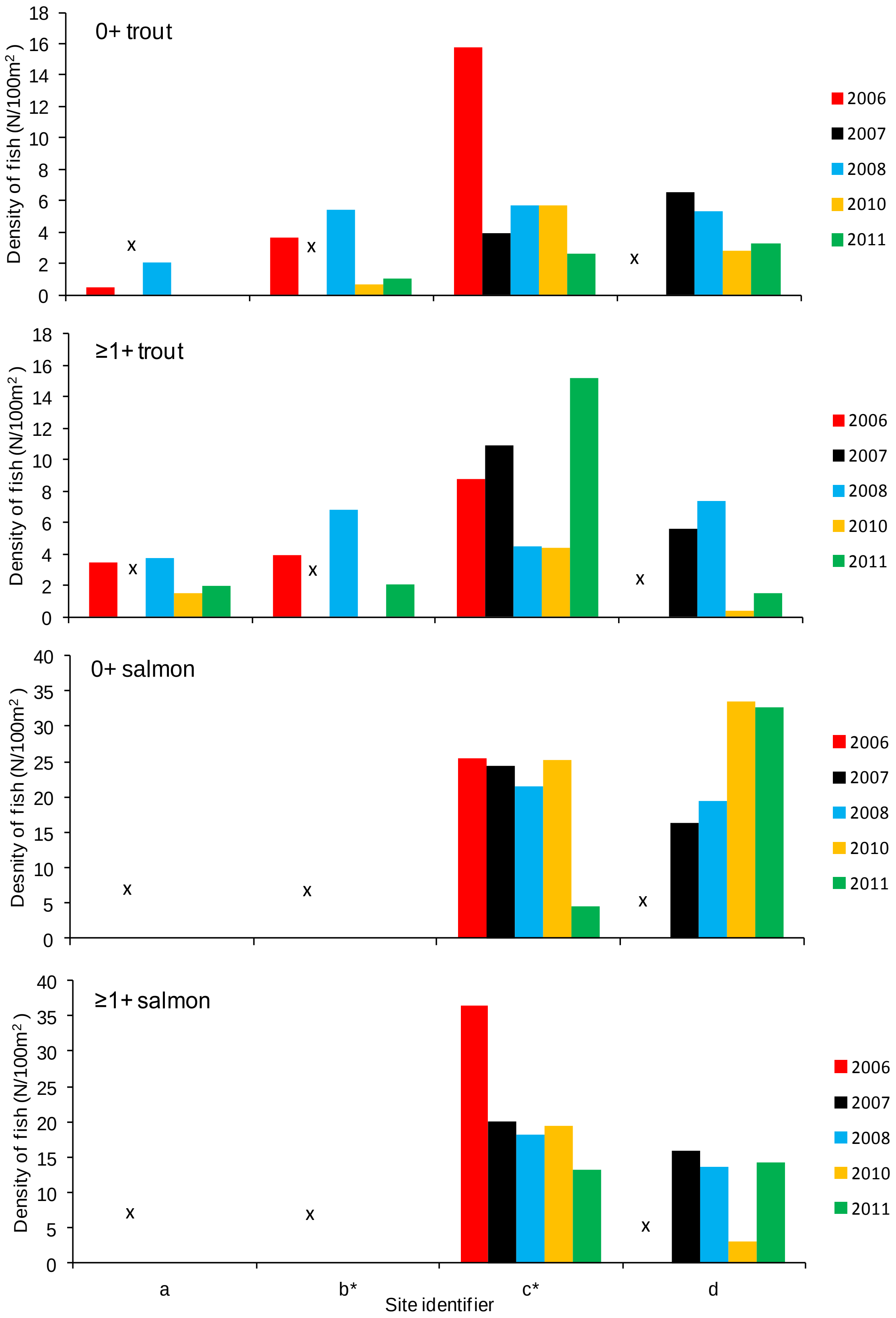

Figure 4.36 Density estimates of trout and salmon at Inverhaggernie between 2006 and 2011. Note, sites a and b were not sampled in 2007 and site d was not surveyed in 2006. No sites were surveyed in 2009. Hydropower scheme was commissioned in 2009. x represents years not surveyed.

Table 4.28 Classification of salmon densities in the lower reaches of Inverhaggernie Burn between 2006 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2006	C	D	D	E	C
	2007	C	D	C	D	C
	2008	C	D	C	D	D
	2010	C	D	D	E	C
	2011	E	E	E	E	E
d	2006	n/s	n/s	n/s	n/s	n/s
	2007	D	D	D	E	D
	2008	C	D	C	D	D
	2010	C	D	C	D	C
	2011	C	D	C	D	C

≥1+ salmon

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2006	A	A	A	A	A
	2007	B	B	B	B	A
	2008	B	B	B	B	B
	2010	C	D	D	D	B
	2011	C	C	C	D	B
d	2006	n/s	n/s	n/s	n/s	n/s
	2007	B	B	B	C	B
	2008	B	B	B	B	B
	2010	E	E	E	E	E
	2011	C	D	D	D	B

4.6.2 Salmonid size structure

Upstream of the intake

Length distributions of trout in the upper reaches associated with the intake varied between years at each site. At site a, only one 0+ trout was caught in 2006 and none were present in 2010 and 2011; ≥1+ trout were caught in the size range 80-200 mm and a narrower size range of 85-175 mm in 2008 (Figure 4.37). ≥1+ trout were caught in low numbers in 2010 and 2011 and in a very narrow size range (Figure 4.37).

Depleted reach

At site b, downstream of the intake, 0+ trout were caught in the size range 50-65 mm and 45-65 mm in 2006 and 2008 respectively, and in a narrower size range in 2010 and 2011 (Figure 4.38). ≥1+ trout were caught in a similar size range (80-170 mm) in 2006, 2008 and 2011 (Figure 4.38).

Length distributions of salmon in the lower reaches of Inverhaggernie Burn were broadly similar between years, albeit in varying numbers, at both sites c and d (Figure 4.39 & 4.40). At site c, upstream of the outfall, 0+ salmon were caught in the size range 45-75 mm during the study period; ≥1+ salmon were caught in the size range 80-130 mm, throughout the study period except in 2010 when ≥1+ salmon were in a narrower size range of 85-115 mm (Figure 4.39). 0+ trout were caught in a similar size range of 40-70 mm in 2006 and 2010, with a slightly narrower size range in other years (Figure 4.41). ≥1+ trout were caught in a wide size range (80-200 mm) in 2007 but a narrow size range in other years (Figure 4.41).

Downstream of the outfall

At site d, 0+ salmon were caught in a narrower size range of 45-70 mm in 2007 and a wider size range of 35-70 mm in other years (Figure 4.40). ≥1+ salmon were caught in the size range 75-125 mm, throughout the study period (Figure 4.40). 0+ trout were caught in the size range 35-75 mm in 2008 and a narrower size range in other years; ≥1+ trout were caught in a similar size range of 85-140 mm in 2007 and 2008 but a narrower size range in other years (Figure 4.42).

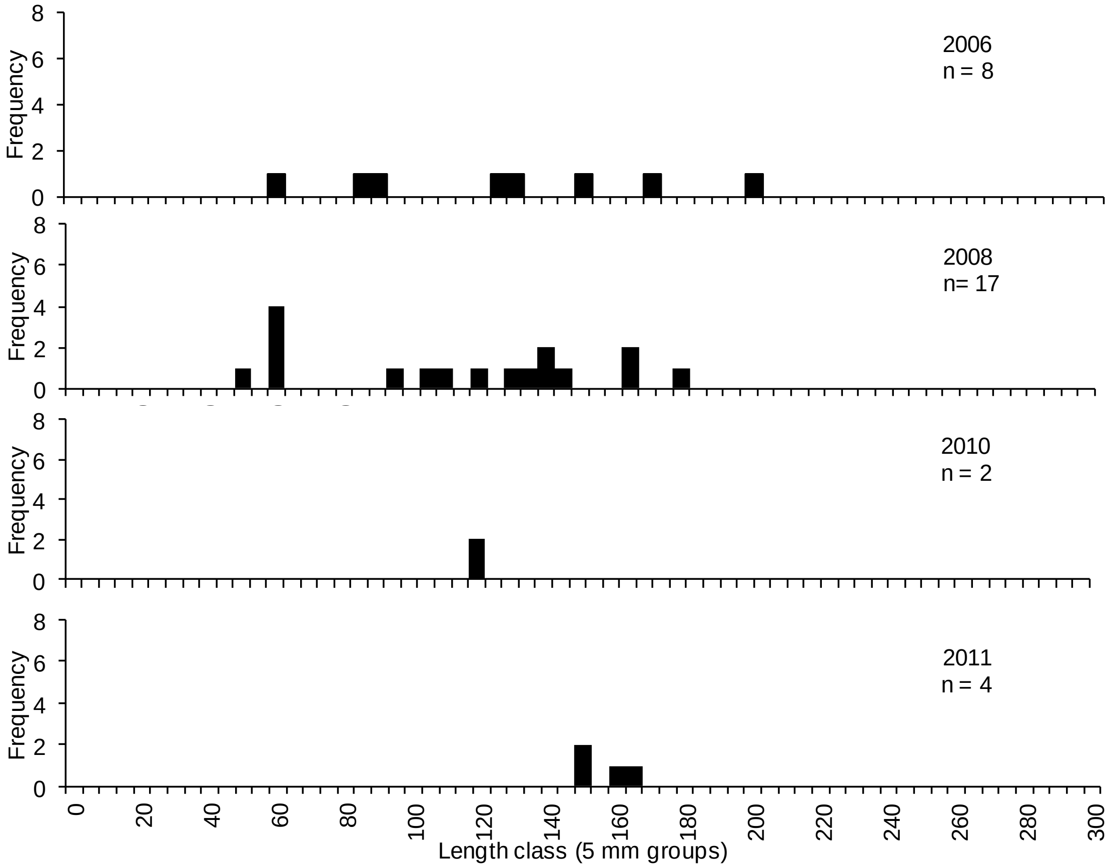

Figure 4.37 Length distributions of trout at site a in Inverhaggernie Burn between 2006 and 2011. Note site a was not surveyed in 2007 or 2009.

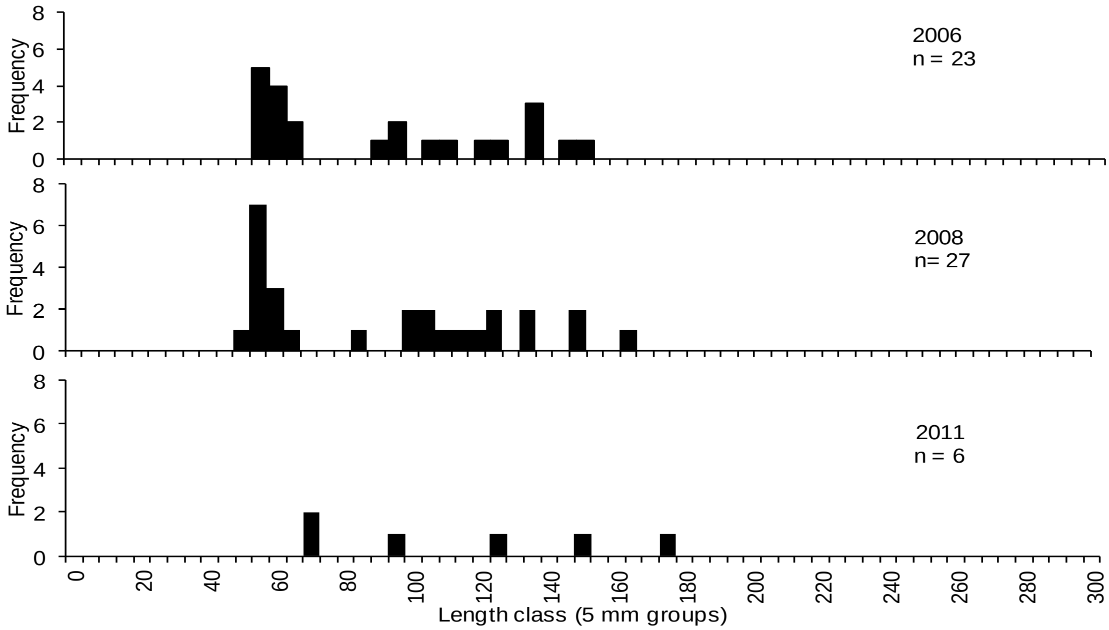

Figure 4.38 Length distributions of trout at site b in Inverhaggernie Burn between 2006 and 2011. Note one trout was caught in 2010 (65 mm) and site b was not surveyed in 2007 or 2009.

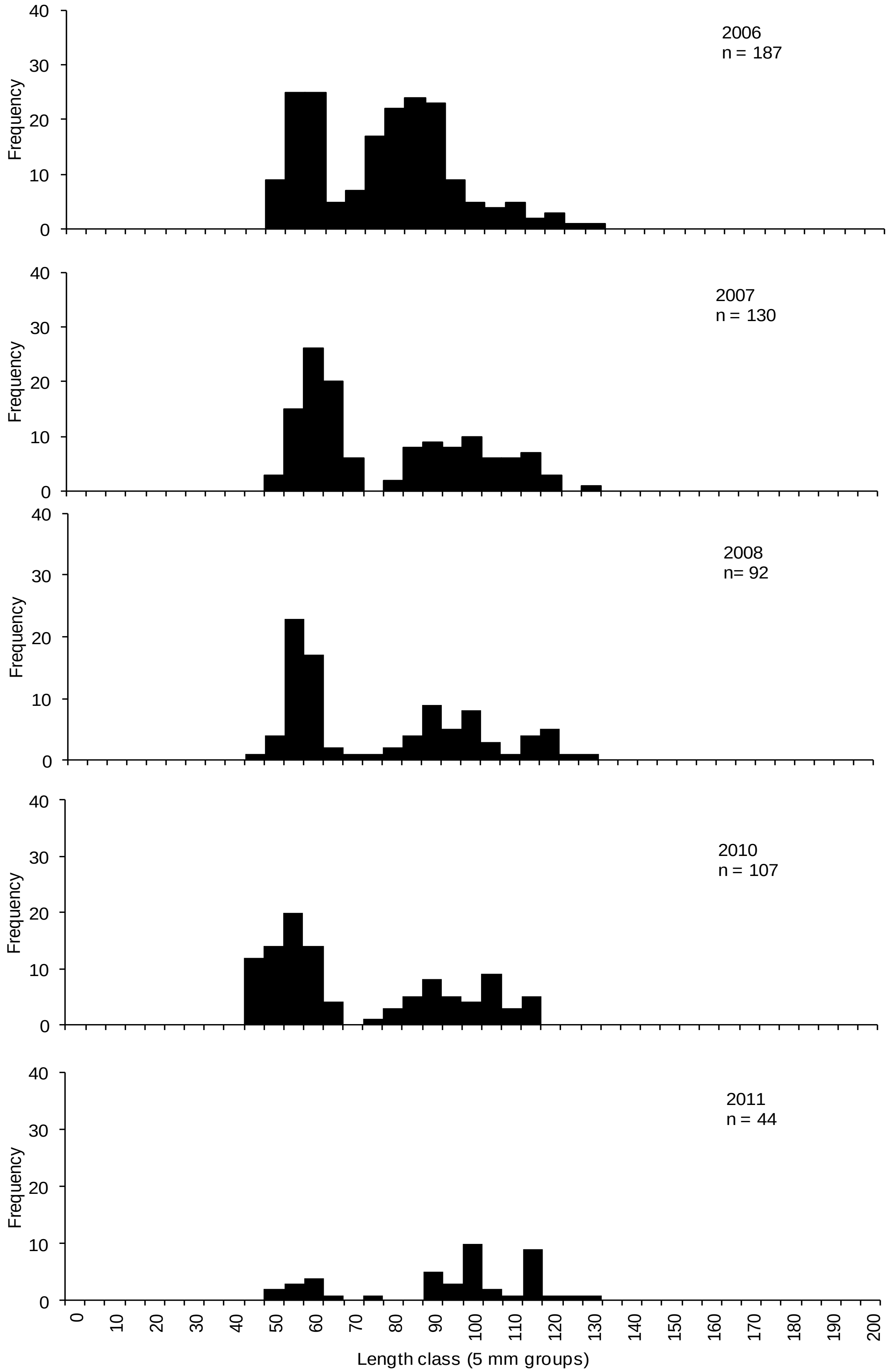

Figure 4.39 Length distributions of salmon at site c in Inverhaggernie Burn between 2006 and 2011. Note site c was not surveyed in 2008.

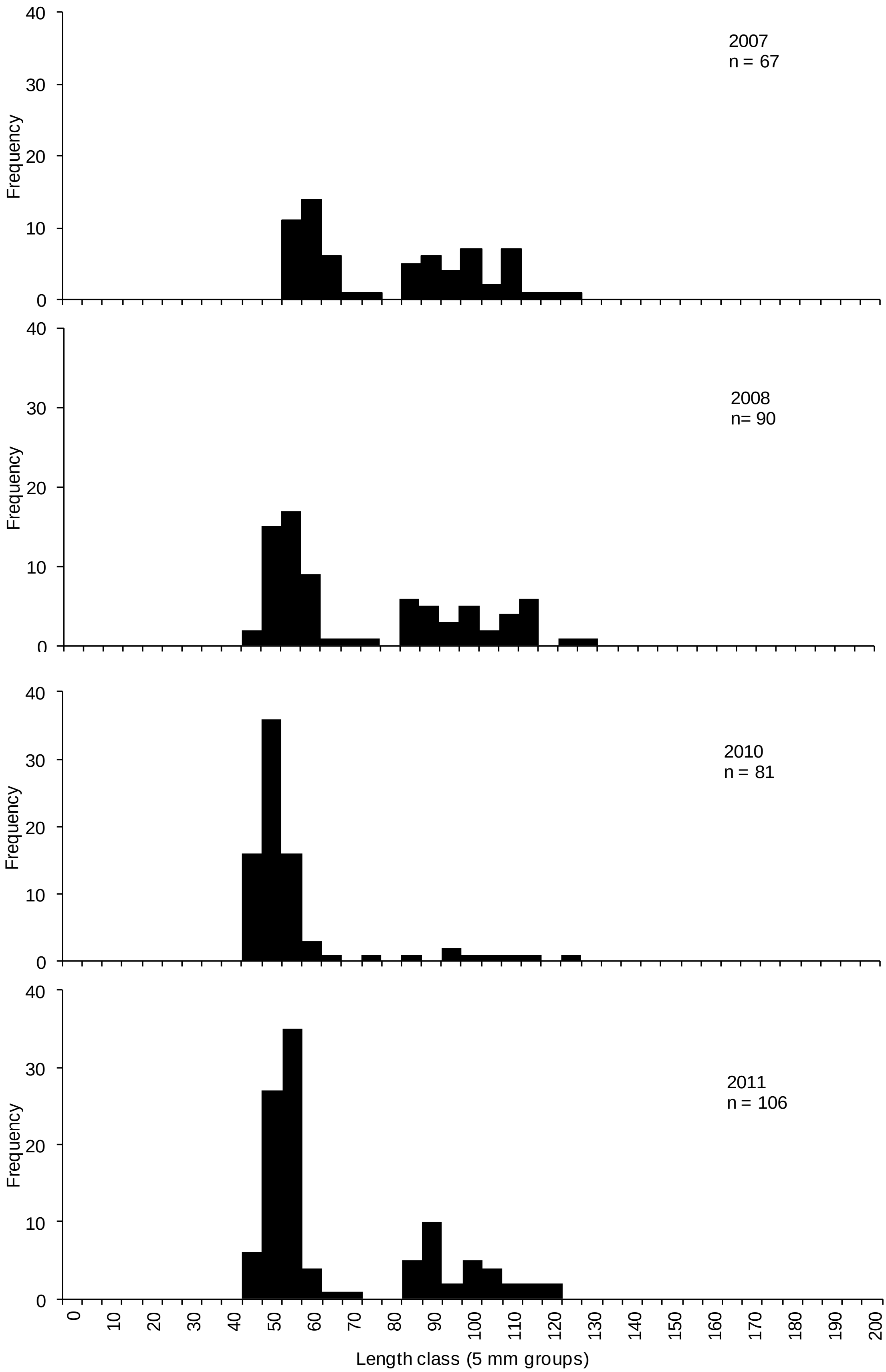

Figure 4.40 Length distributions of salmon at site d in Inverhaggernie Burn between 2007 and 2011. Note site d was not surveyed in 2006 or 2009.

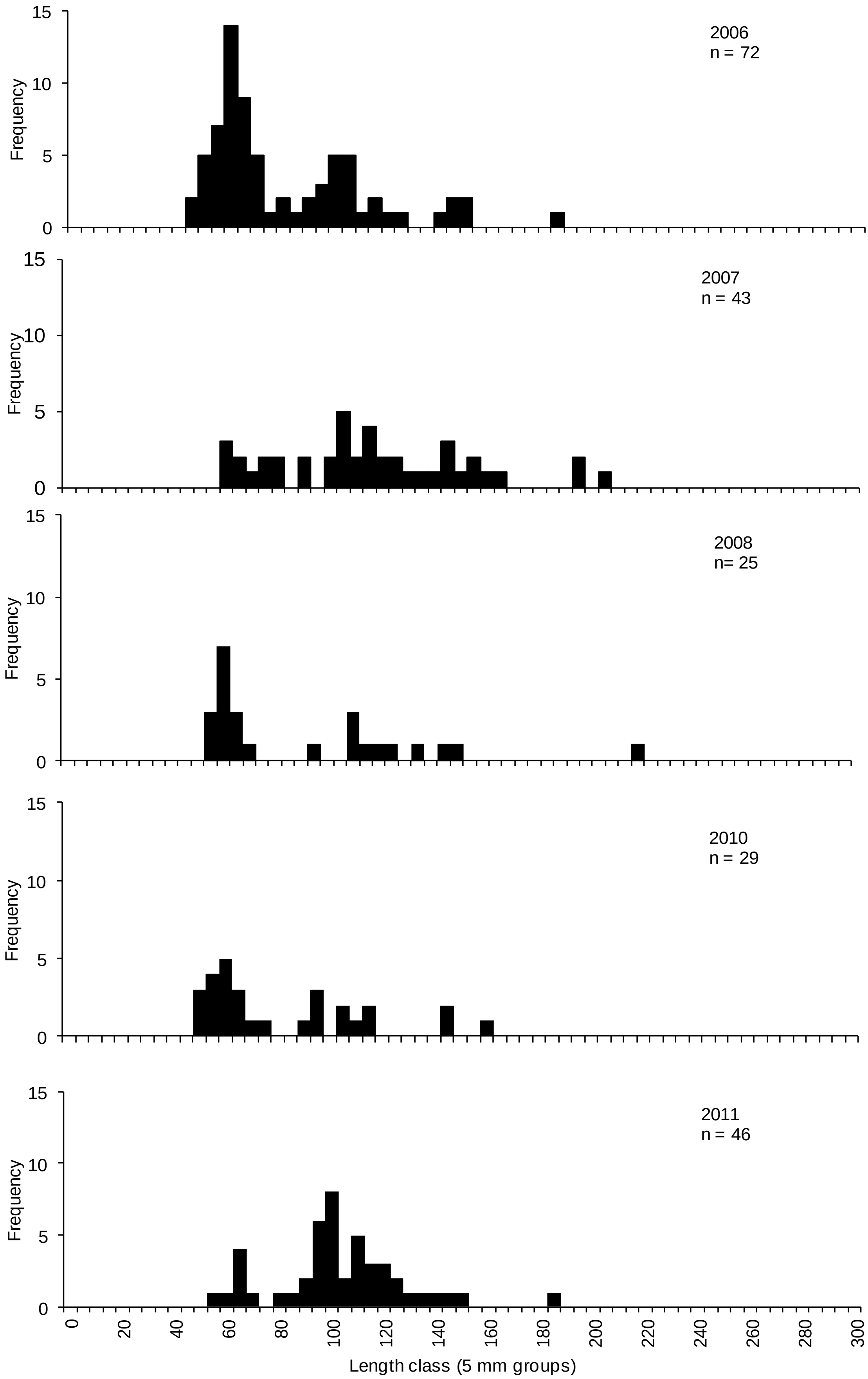

Figure 4.41 Length distributions of trout at site c in Inverhaggernie Burn between 2006 and 2011. Note site c was not surveyed in 2008.

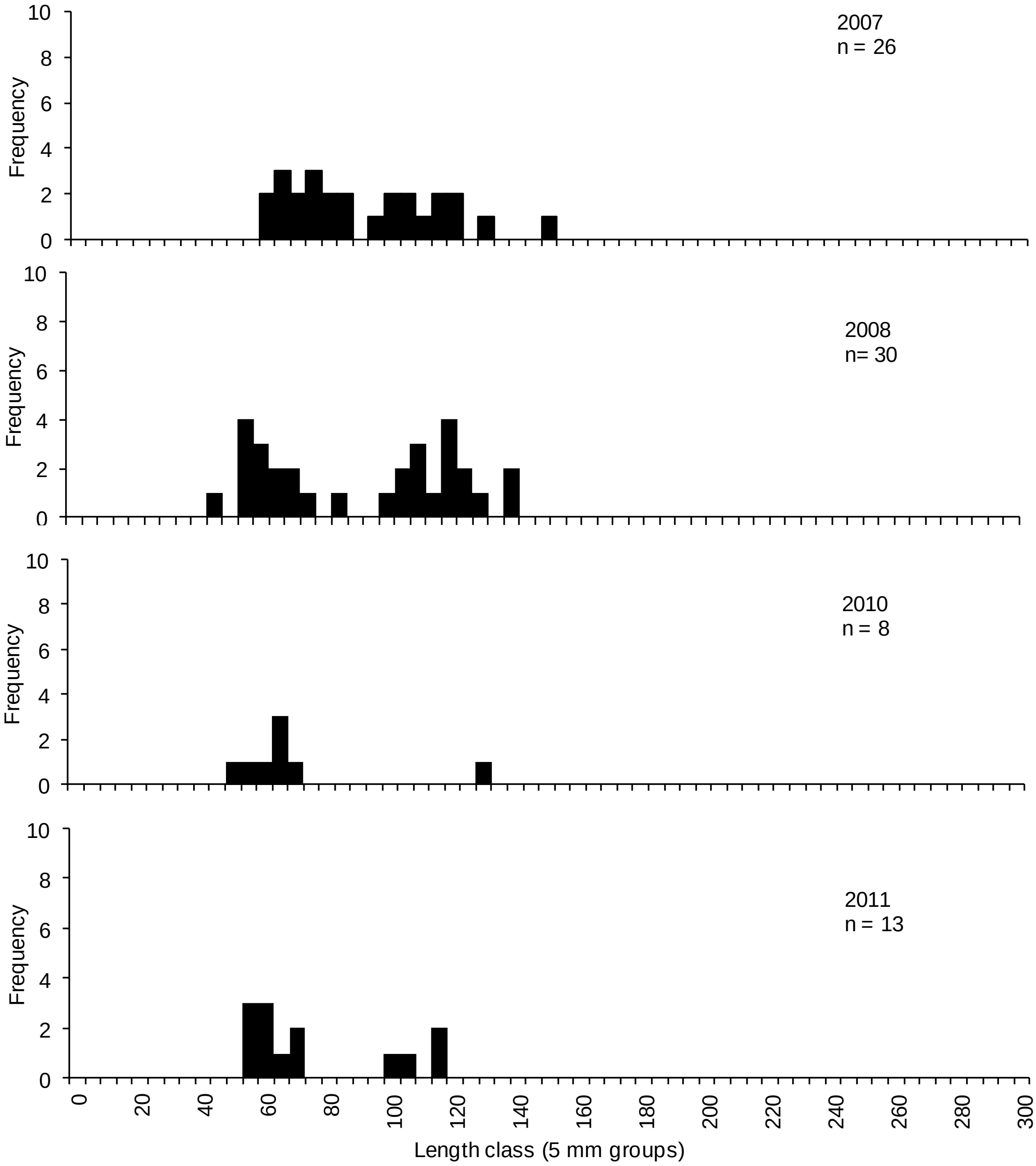

Figure 4.42 Length distributions of trout at site d in Inverhaggernie Burn between 2006 and 2011. Note site d was not surveyed in 2006 or 2009.

4.6.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was commissioned were compared to the same information collected following commissioning at two key sites in the lower reaches of Inverhaggernie Burn. For the analysis habitat data collected in 2008 were utilised with fisheries data collected from 2006-2008 and compared to habitat data collected in 2011 utilised with fisheries data collected from 2010 and 2011 (Tables 4.29 & 4.30). The data allowed comparison between one site in the abstracted reach and one site downstream of the abstracted reach.

Table 4.29 HABSCORE outputs for site c (abstracted reach) before (2006-2008) and after (2010-2011) the Inverhaggernie Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2006-2008	68.0	25.27	14.47	4.08	51.34	1.75	0.23	13.24	0.55
2010-2011	28.5	10.59	72.83	17.27	307.14	0.15	0.02	1.23	-1.90
≥1+ salmon									
2006-2008	68.5	25.43	6.97	1.97	24.73	3.65	0.56	23.57	1.29
2010-2011	43.2	16.05	4.39	1.21	15.95	3.65	0.55	24.11	1.29
0+ trout									
2006-2008	20.5	7.60	23.25	5.89	91.86	0.33	0.05	2.25	-1.10
2010-2011	10.9	4.05	17.91	4.53	70.80	0.23	0.03	1.55	-1.47
≥1+ trout (<20 cm)									
2006-2008	21.0	7.80	7.71	1.75	33.99	1.01	0.16	6.22	0.09
2010-2011	22.8	8.46	2.96	0.65	13.42	2.86	0.45	18.11	1.05
≥1+ trout (>20 cm)									
2006-2008	1.0	0.37	0.30	0.09	0.94	1.25	0.39	4.00	0.22
2010-2011	0	0	0.34	0.11	1.09	1.10	0.34	3.53	0.10

Table 4.30 HABSCORE outputs for site d (control) before (2006-2008) and after (2010-2011) the Inverhaggernie Burn hydropower scheme was commissioned. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL. column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2006-2008	40.6	16.67	24.27	6.84	86.07	0.69	0.09	5.21	-0.37
2010-2011	91.0	32.96	117.28	26.38	521.39	0.28	0.03	2.48	-1.27
≥1+ salmon									
2006-2008	33.5	13.77	7.27	2.04	25.81	1.90	0.29	12.26	0.64
2010-2011	17.4	6.32	4.96	1.36	18.07	1.27	0.19	8.44	0.24
0+ trout									
2006-2008	14.0	5.76	28.20	7.18	110.83	0.20	0.03	1.40	-1.60
2010-2011	7.9	2.88	16.91	4.24	67.36	0.17	0.02	1.18	-1.77
≥1+ trout (<20 cm)									
2006-2008	14.3	5.87	3.15	0.71	14.00	1.86	0.30	11.55	0.62
2010-2011	2.0	0.72	1.12	0.24	5.09	0.65	0.10	4.12	-0.43
≥1+ trout (>20 cm)									
2006-2008	0.0	0.00	0.14	0.04	0.44	3.00	0.94	9.57	1.09
2010-2011	0	0	0.11	0.03	0.36	3.26	1.02	10.42	1.18

The observed density of 0+ salmon at site c, upstream of the outfall, prior to commissioning of the hydropower scheme (2006-2008 data) was higher than predicted by the HQS suggesting better population than expected but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.29). The observed density of 0+ salmon following commissioning of the hydropower scheme (2010-2011 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.29). The observed densities of ≥1+ salmon at site c both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.29). The observed densities of 0+ trout at site c both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CL was >1 for both scenarios therefore the observed populations were not significantly lower than expected (Table 4.29). The observed densities of ≥1+ (<20 cm) trout at site c both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.29). The observed density of ≥1+ trout (>20 cm) at site c prior to commissioning of the hydropower scheme (2006-2008 data) was higher than predicted by the HQS suggesting better population than expected but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.29). The observed density of ≥1+ trout (>20 cm) following commissioning of the hydropower scheme (2010-2011 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.29).

The observed densities of 0+ salmon at site d, downstream of the outfall, both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CL was >1 for both scenarios therefore the observed populations were not significantly lower than expected (Table 4.30). The observed densities of ≥1+ salmon at site c both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were higher than predicted (HQS), suggesting better populations than expected but the HUI lower CLs were <1 in both scenarios therefore the observed populations were not significantly higher than expected (Table 4.30). The observed densities of 0+ trout at site d both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CL was >1 for both scenarios therefore the observed populations were not significantly lower than expected (Table 4.30). The observed density of ≥1+ trout (>20 cm) at site d prior to commissioning of the hydropower scheme (2006-2008 data) was higher than predicted by the HQS suggesting better population than expected but the HUI lower CL was <1 therefore the observed population was not significantly higher than expected (Table 4.30). The observed density of ≥1+ trout (>20 cm) following commissioning of the hydropower scheme (2010-2011 data) was lower than predicted (HQS) suggesting poorer populations than expected, but the HUI upper CL was >1 therefore the observed population was not significantly lower than expected (Table 4.30). The observed densities of ≥1+ trout (>20 cm) at site d both prior to and following commissioning of the hydropower scheme (2006-2008 and 2010-2011 data) were lower than predicted (HQS), suggesting poorer populations than expected but the HUI upper CL was >1 for both scenarios therefore the observed populations were not significantly lower than expected (Table 4.30).

4.6.4 Salmonid growth rates

Upstream of the intake

At site a back-calculated length of trout at age 1 ranged between 55 and 69 mm, with the slowest growth in 2006 and fastest in 2007 (Table 4.31). Back-calculated length of trout at age 2 ranged between 108 and 114 mm, with the slowest growth in 2006 and fastest in 2009.

Depleted reach

At site b back-calculated length of trout at age 1 ranged between 58 and 73 mm, with the slowest growth in 2006 and fastest in 2007 (Table 4.31). Back-calculated length of trout at age 2 ranged between 105 and 123 mm, with the slowest growth in 2004 and fastest in 2009. At site c back-calculated length of trout at age 1 ranged between 53 and 62 mm, with the slowest growth in 2005 and fastest in 2004, 2008 and 2010 (Table 4.31). Back-calculated length of trout at age 2 ranged between 108 and 127 mm, with the slowest growth in 2004 and 2006 and fastest in 2003. Back calculated length of trout at age 3 ranged between 156 and 157 mm, and at age 4 was 188 mm in 2004 (Table 4.31). At site c back-calculated length of salmon at age 1 ranged between 42 and 64 mm, with the slowest growth in 2004 and fastest in 2010; back-calculated length of salmon at age 2 ranged between 84 and 90 mm (Table 4.32).

Table 4.31 Back-calculated lengths at age of trout in the upper and lower reaches of Inverhaggernie Burn. Note no 0+ trout were captured in 2011 at site a. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))			
		1	2	3	4
a	2004	61±7(2)	111±8(2)		
	2005	56±11(4)			
	2006	55±10(5)	108±23(5)		
	2007	69±11(5)			
	2009	65±7(6)	114±7(4)		
b	2004	61±7(4)	105±7(4)		
	2005	63±9(8)			
	2006	58±3(3)	112±3(3)		
	2007	73±10(9)			
	2009	68(1)	123(1)		
	2010	67±15(3)			
	*2011	67±1(2)			
c	2003	60(1)	127(1)	157(1)	
	2004	62±7(5)	108±6(5)	156(1)	188(1)
	2005	53±11(13)			
	2006	59±10(13)	108±11(3)		
	2007	57±8(8)			
	2008	62±2(2)	115±8(2)		
	2009	60±6(11)	112±14(3)		
	2010	62±11(16)			
	*2011	62±5(7)			
d	2004	41±4(2)	94±1(2)		
	2005	57±9(10)			
	2006	52±5(9)			
	2009	67(1)			
	2010	65±7(4)			
	*2011	59±6(9)			

Table 4.32 Back-calculated lengths at age of salmon in the lower reaches of Inverhaggernie Burn. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))	
		1	2
c	2004	42±5(6)	84±4(6)
	2005	45±8(11)	
	2008	46±3(4)	90±5(4)
	2009	59±8(18)	
	2010	64±11(13)	
	*2011	57±6(11)	
d	2005	47±10(6)	78±8(3)
	2006	48±9(15)	93±9(4)
	2007	66±11(12)	
	2008	48(1)	90(1)
	2009	64±10(8)	
	2010	63±9(13)	
	*2011	51±4(74)	

Downstream of the outfall

At site d back-calculated length of trout at age 1 ranged between 41 and 67 mm; with the slowest growth in 2004 and fastest in 2009; back-calculated length of trout at age 2 was 94 mm in 2004 (Table 4.31). At site d back-calculated length of salmon at age 1 ranged between 47 and 66 mm; with the slowest growth in 2005 and fastest in 2007 (Table 4.32). Back-calculated length of salmon at age 2 ranged between 78 and 93 mm, with the slowest growth in 2005 and fastest in 2006 (Table 4.32).

4.6.5 Eel

Eels were captured in the upper and lower reaches associated with the intake and outfall. Numbers of eels captured at sites a-c varied marginally between survey years and data suggest the upper and lower reaches of Inverhaggernie Burn are used by small numbers of eels (Table 4.33). Despite slight variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers with the hydropower scheme operational.

Table 4.33 Number of eel captured at survey sites in the upper and lower reaches of Inverhaggernie Burn between 2006 and 2011. No eels were caught at site d throughout the study period, n/s not surveyed.

Year	Site identifier		
	a	b	c
2006	1	3	3
2007	n/s	n/s	2
2008	0	4	1
2009	n/s	n/s	n/s
2010	0	0	0
2011	0	0	4

4.6.6 Wetted widths

A paired two sample *t*-test was performed on mean wetted widths (collected during HABSCORE) at site c, as this was located within the abstracted reach. Essentially, the test was used to distinguish if there was any difference between wetted widths each time HABSCORE was performed to ensure comparison of density estimates over the study period. The 95% confidence limits (-2.395, 0.888) enclosed zero with a p value of 0.345, n = 27; therefore there was no significant difference between wetted widths pre and post abstraction, confirming comparability of density estimates.

4.7 Discussion

At Kinnaird Burn, Keltney Burn and Innerhadden Burn, salmon and trout populations varied over the study period. Densities of fish varied both within and outside the impacted reach, thus natural fluctuations made it difficult to detect any impacts specifically associated with commissioning of the hydropower schemes. Although juvenile salmon were absent in Kinnaird Burn throughout the study period, this was exclusively a problem with damage to a fish pass in the lower reaches of the burn. HABSCORE data, however, suggest that should salmon be able to ascend to the lower reaches of Kinnaird Burn in the future there would be suitable spawning and juvenile habitat available.

Reduction in density estimates

At Rottal Burn and Inverhaggernie Burn, a reduction in ≥1+ and 0+ salmon density respectively, was observed in the depleted reach, post hydropower commissioning. These declines were not reflected in the fish densities located at the control site downstream of the abstracted reach and thus suggest an impact of flow regulation. Reductions in 0+ trout density were also observed upstream of the intakes at both Rottal Burn and Inverhaggernie Burn. While sites upstream of the intake act as a control for flow regulation, they are subject to other impacts associated with the hydropower scheme; possible reasons for the decline in 0+ fish upstream of intakes are discussed below. It is important to note that the potential impacts of hydropower schemes are different upstream of the intake and in the depleted reach. The main impact in the depleted reach is a reduction in the amount of water, leading to associated changes in habitat including important spawning/nursery areas. The main impact upstream of the intake is reduced access because of the intake weir, which may be exacerbated by the reduction in the amount of water downstream. Therefore, impacts can be observed upstream of an intake (barrier effect), upstream and downstream of an intake (barrier and abstraction effect) and downstream of an intake (abstraction effect).

The Rottal Burn hydropower scheme commenced abstraction in December 2008, ≥1+ salmon densities subsequently reduced quite dramatically at site e located in the depleted reach, between 2009 and 2010. ≥1+ salmon populations did not decline at the downstream control site (g), instead densities increased between 2009 and 2010 to some of the highest recorded densities of ≥1+ salmon over the study period. While ≥1+ salmon densities at site g (control site) fluctuated between 2009 and 2011 (post hydropower abstraction), all densities recorded in this period were still higher than those recorded in 2007 (pre hydropower abstraction). ≥1+ salmon densities at site f, located in the depleted reach, also increased between 2009 and 2010, mirroring the trend observed at control site g. However, densities observed in 2011 (post abstraction) were much lower than those observed in 2007 (pre abstraction) and the lowest recorded at this site over the study period. This general declining trend between

2007 (pre abstraction) and 2011 (post abstraction) was also observed at site e (located in the depleted reach) but the opposite of that observed at site g (control site). It should be noted however, that the general pattern observed in densities at site f (depleted reach) and g (control site) are mirror images of one another and display the same increasing trend between 2009 and 2010 and subsequent decrease between 2010 and 2011. Thus fluctuations in densities at site f (depleted reach) are considered in response to natural variability. The increasing trend between 2009 and 2010 was not observed at site e (depleted reach), and ≥1+ salmon densities reduced from 36/100 m^2 (2009) to 3/100 m^2 (2010). Although this decline could be a natural phenomenon, given that it was only observed in one age class at one site in the depleted reach, it is also possible that abstraction has had an impact at this particular site. Indeed, there are possible reasons that could explain the reduction of ≥1+ salmon at site e, discussed below. It is highlighted that there was no significant difference in wetted widths before and after each of the hydropower schemes commenced abstraction, indicating that the abstraction of water was not extreme (at least not during the surveys undertaken within this study). The implications of this are that any increases in density in the depleted reach are not because the fish are now occupying a smaller area of water, but because there are more fish (not more fish per unit area). Conversely, any reduction in density must have been caused by there being fewer fish, because the area of water was not significantly different. It should be noted that abstraction does not necessarily have to have been severe during the surveys to have had an impact; it could have been severe during other, possibly critical times of the year, such as the migration/spawning season.

It is important to note firstly that ≥1+ salmon could have moved considerable distances from areas that are uninfluenced by the scheme, and secondly that ≥1+ fish would have been spawned/hatched before the Rottal Burn hydropower scheme was commissioned. It is therefore not possible to detect an impact of the hydropower scheme on spawning success using this age class. Changes in 0+ fish densities however, can be used as an indicator of spawning success. 0+ trout and salmon densities at site e generally increased between 2009 and 2011. However, a general decreasing trend was observed in 0+ salmon densities at site f, also located in the depleted reach, between 2009 and 2011, but this was mirrored at the control site downstream, indicating natural fluctuations.

While densities of 0+ fish did not decline at site e, this is not to say that a reduction of flow did not impact upon ≥1+ salmon directly by altering the habitat and thus restricting their ability to utilise the area. Unfortunately HABSCORE data were not collected at this site and so density estimates in relation to the habitat available before and after the scheme was commissioned cannot be analysed. Nevertheless, it is a possibility that a reduction in water depth and/or velocity or perhaps a change in substratum composition reduced the habitat suitability for ≥1+ fish. Consequently, the site located downstream of the abstracted reach could have, in comparison to that at site e, been more suitable, due to higher flows, supporting the reduced densities of ≥1+ fish at site e. Indeed it is documented that adult fish are more likely to move into temporary habitats to compensate for reductions in the quality or availability of habitat (Kraft, 1972). At site g, located further downstream, outside of the abstracted reach, ≥1+ salmon densities increased between 2009 and 2010 as ≥1+ salmon densities decreased at site e. While it is a possibility that ≥1+ salmon could have moved from the depleted reach at site e to site g, where it is likely that flows were higher as it is located outside of the abstracted reach, an increase in ≥1+ salmon at site g in response to natural annual variation in recruitment must also be considered. Indeed, there was also an increase in ≥1+ salmon at site f (located within the abstracted reach). From personal observations, the terrain at site e is particularly steep in comparison to that of sites located further downstream, and so it could be considered that reduced flow caused by abstraction has made negotiating this terrain more difficult for the larger individuals (≥1+ fish). It is more difficult for larger fish to move upstream when flows are reduced as could be experienced in the depleted reach of Rottal Burn. Although discharge data were unavailable to indicate the exact amount of water that was abstracted, upstream areas can become inaccessible when discharges are low (Gibbins *et al.,* 2008). As HABSCORE data were not collected at this site, wetted widths before/after abstraction began could not be compared. However, no significant difference in the mean wetted width at site f (located further downstream), before and after the scheme became operational was found, suggesting abstraction could not have been that severe (during the time of surveys) and was unlikely to affect accessibility to site e of ≥1+ fish. It must also be taken into account that ≥1+ salmon may indeed have hatched at site e and so would not necessarily need to negotiate the terrain, however, this information is unknown and outside the scope of this study and therefore all possibilities must be explored. Ultimately, it is possible that a combination of factors including reduced habitat suitability and reduction of water has affected ≥1+ salmon at site e, though the former is considered to be the more likely driver of the decline in densities.

The theory of reduced flow disrupting a river's longitudinal connectivity could, in part, further explain why the overall number of trout dramatically reduced at Rottal Burn, site a, upstream of the intake weir. A consecutive annual decline, post hydropower commissioning, was observed in trout numbers (both age groups combined) from 109 in 2009 (76/100 m^2), 59 in 2010 (52/100 m^2) to only 24 in 2011 (21/100 m^2). Densities pre-commissioning were similar to those recorded in 2010 at 53/100 m^2 (2006) and

55/100 m^2 (2007). While it is possible that natural variation may account for the fluctuations in densities given that densities in 2010 (post-commissioning) were similar to those in 2006 and 2007 (pre-commissioning) there are two other factors that must be explored. Firstly, from 2009 onwards, adult fish may (some brown trout are resident) have had to negotiate a barrier that was built to accommodate the hydropower scheme in order to access habitat at site a. The presence of the weir however may have prevented returning adults from accessing this upstream site. Although resident brown trout do not migrate out to sea, they can move between reaches of a river. Thus, it is likely that resident brown trout may have passed between sites a and b before the intake weir was built but were perhaps inhibited from doing so when the intake weir was built. Secondly, reduced flow in the stretch downstream of the weir (during abstraction) could have further increased difficulties for those fish attempting to pass the obstacle. While a residual flow must be maintained in the depleted stretch of river, the general reduction in numbers of fish caught at site a could be in part due to insufficient flow over the weir, perhaps preventing fish from negotiating the barrier. It is highlighted again, however, that not all adults emigrate after spawning, thus residential brown trout would not need to negotiate the barrier. Although the off-take weir built at Rottal Burn (Figure 3.6) is considered small at <1 m, it has been discussed that weirs with a head loss of <5 m can have significant effects on flow and temperature regimes, sediment transport, biogeochemistry, animal movements and stream habitat (Larinier, 2001; Hart *et al.,* 2002), thus modifications of flow caused by such barriers can also alter the structure of communities and the function of river ecosystems (Baumgartner, 2007). Radio-tracking studies revealed that weirs with a head loss of 0.5 m, of similar dimensions to the weirs in this study, can delay the movement and passage of Atlantic salmon and sea trout, while weirs of 1.2 m head loss can be impassable obstacles depending on flow and water temperatures (Gerlier & Roche, 1998). It is important to recognise that the ability of fish species to bypass weirs and other obstructions is dependent on the hydraulic conditions at the structure. Partial barriers can become a full barrier if environmental conditions are altered; swimming capabilities are reduced at lower temperatures meaning even small obstacles may be difficult to ascend at certain water temperatures. Gerlier & Roche (1998) reported waterfalls that were not passed by upstream migrating Atlantic salmon when water temperatures dropped below 10^{o}C. Therefore, while the Rottal Burn intake weir is considered a small barrier, it could have become impassable if temperatures were low during the autumn/winter when adults were migrating upstream; adults may have been restricted from accessing site a, resulting in reduced recruitment and thus a decline of 0+ fish. These considerations could equally be applied to explain the reduction of 0+ trout at site a, upstream of the intake weir (Figure 3.10) at Inverhaggernie Burn in 2010 and 2011. Access to site a, was perhaps inhibited by the presence of the weir, and thus adult fish were unable to utilise the spawning habitat resulting in a loss of recruitment the subsequent years. Additionally, as site a is immediately upstream of the intake, the ‘ponding’ effect could have caused a change in habitat, reducing the suitability of this site for brown trout and resulting in a reduction of 0+ densities.

The Inverhaggernie Burn hydropower scheme commenced abstraction in October 2009, 0+ salmon populations at site c located within the depleted reach reduced dramatically between 2010 and 2011. These data suggest a possible impact of flow regulation as the decreasing trend was not reflected in the fish population at site d located outside of the depleted reach and occurred after the hydropower scheme had been commissioned.

0+ fish in 2011 would have been spawned/hatched after the Inverhaggernie Burn hydropower scheme was commissioned (October 2009), and can therefore indicate potential effects of abstraction, such as reduced flows and habitat alteration, on spawning success. The reduction in salmon densities at this site could indeed be in response to reduced flows following abstraction of the hydropower scheme. Indeed, Ugedal *et al.* (2008) identified a reduction of 80% in the densities of juvenile Atlantic salmon (followed by a reduction of returning adult salmon) following river regulation from pre-regulation levels in 1992 to minimum levels in 1996, thus reflecting the response of fish to decreased water flows. A number of authors have reported reduced stream fish populations in response to reduced flows (Elliott *et al.,* 1997; Magoulick & Kobza, 2003; Hakala & Hartman, 2004) or drying disturbances (Davey & Kelly, 2007). By contrast, Covich *et al.* (2003) found low-flow periods, such as those likely to be experienced in a depleted stretch of river, like Inverhaggernie Burn, altered fish communities by ultimately reducing water depth and wetted area, causing increased fish density. This is known as a false positive because the numbers of fish have not actually increased, but densities appear to have done so relative to the reduced volume of water. Increase fish densities in a reduced amount of space can in turn can lead to decreased production (Keaton *et al.,* 2005; Davey & Kelly, 2007) because of density dependent interactions (Magoulick & Kobza, 2003). Increased fish densities were observed in the depleted reach of Keltney Burn (discussed later in this section), however it should be noted that there was no significant difference between wetted widths before and after hydropower commissioning, indicating this was a true increase in densities.

Fish can also become stranded on gravel bars or trapped in off-channel habitats during flow decreases (Bunn & Arthington, 2002) which are undoubtedly experienced in the depleted stretches of a river. Jowett *et al.* (2005) noted that years with the lowest flows (all natural) had the most substantial reductions in fish abundances (juveniles and adults). It is considered that adult fish need high flows in order to begin their migration upstream (Chapter 2), and will move on the receding end of a spate. Thus it is possible that years that experience low flows may deny adult fish their environment cues and consequently result in a reduction of adult fish and subsequently, 0+ fish. In addition, Bradford (1997) expressed concerns that larval salmonids are at risk of being stranded in the substratum if flow reductions occur in winter. While Inverhaggernie Burn abstraction rates are unavailable, during periods of abstraction the flow in the river between the intake and outfall will be reduced. Consequently, the factors discussed above relating to flow decreases could all contribute to reduced 0+ salmon densities at site c. Furthermore, abstraction rates at critical periods in autumn/winter 2010 when adult salmon moved upstream to spawn may have been higher than in autumn/winter 2009 thus impeding their movement; but unfortunately the data to assess abstraction in relation to flow were not available for analysis. That said, it should be noted that if abstraction were to be higher, it is likely that river discharge would also have been higher; generally abstraction increases with increasing river discharge.

Equally, the decline could be related to the presence of a weir that was placed at the outfall. Unlike the other schemes, the depleted reach at Inverhaggernie Burn was also impacted by a possible migration barrier. Prior to construction of the scheme, the reach adjacent to the outfall was characterised by a shallow gradient with unimpeded passage for migratory salmonids. However, when carrying out fish surveys in September 2010 the reach had an artificial weir constructed of natural substrata in place, possibly to facilitate the outfall discharge and reduce scouring (Figure 4.43).

The artificial weir was a substantial structure when photographed and may have restricted adult salmon movements at critical flows. It is not clear if the weir was put in place prior to abstraction commencing (November 2009) or at a later date; no consent had been granted for this structure confirmed by SEPA and as such there is no record available to aid with interpretation of the results. If the weir was put in place after the autumn/winter of 2009 then adult salmon movement would not have been impeded hence the good 0+ densities recorded in 2010. However, if the weir was in place prior to the autumn/winter flows of 2010 then this may have restricted adult salmon passage, possibly resulting in the low 0+ densities found in 2011.


(a)


(b)


(c)

Figure 4.43 Outfall location prior to installation (a), in September 2010 with retaining weir (b) and September 2011 with retaining weir (c).

In September 2011 when the site was visited, the weir appeared to have been partially washed away and therefore would be a less restrictive barrier to adult salmon movements in autumn/winter 2011. However, it is not clear when the weir was washed away and this could equally have been prior to the autumn/winter of 2010 and so in theory would have been less of a barrier to adult salmon movements and thus the reason for the low 0+ salmon density may not be related to the weir. However, it is reinforced that 0+ salmon densities downstream of the weir (site d) were higher in 2010 and 2011 post hydropower commissioning than those recorded in 2006-2008, pre hydropower commissioning. It is therefore possible, adult salmon could not negotiate the weir to access site c and so utilised site d for spawning resulting in the increased densities of 0+ salmon. It was also considered that the presence of the weir might have altered the habitat upstream of the outfall making site c less suitable for fry and thus supporting the decline in densities, however, from personal observation the habitat both upstream and downstream of the weir has not changed. This is further supported by the HABSCORE outputs, which confirms no deviation in habitat before and after commissioning of the scheme.

Habitat complexity increases with water depth, water velocity and cover (Gorman & Karr, 1978; Schlosser, 1982; Felley & Felley, 1987), resulting in increases in the richness of aquatic fauna. Reductions in flow, like those in the depleted reaches, can therefore result in shallow areas and side channels drying out, thus reducing the amount of potential spawning and/or nursery habitat. Indeed, Anderson *et al.* (2006) characterised reaches with artificially reduced discharge (depleted flows) as having slower velocities, increased water temperatures and shallower habitats compared with upstream and downstream of the depleted reach. Generally, however, HABSCORE analysis of observed and predicted densities in this study, showed no deviation for both pre and post hydropower commissioning scenarios, indicating no obvious impact of flow regulation on fish population predictions or habitat quality. Although HABSCORE analysis revealed 0+ salmon densities at site c (abstracted reach) in Inverhaggernie Burn were higher than predicted pre hydropower commissioning but lower than predicted post hydropower commissioning, a paired *t*-test revealed there was no significant difference between fish density at this site before and after commissioning of the hydropower scheme. However, it should be noted that the results are based on a small number of samples due to the length of study period and therefore high variation may be present. This does not necessarily imply that the density estimates were inaccurate but rather that even if there was a substantial difference in 0+ salmon densities before and after hydropower commissioning, it may not be statistically significant due to the high inter-annual variations. It should also be noted that, although this could be a genuine result, it could perhaps be an artefact of HABSCORE and *t*-tests doing different things. The *t*-test simply compared mean density before and after the hydropower scheme was commissioned, whereas HABSCORE predicted the density based upon the habitat, which may have changed. So, in theory, the density could be identical before and after the hydropower scheme was commissioned, but higher than predicted before and lower than predicted after if the habitat had changed. Essentially, however, the habitat at site c post hydropower abstraction was no different to that observed before hydropower abstraction and could have supported higher densities of 0+ salmon than there were, based on HABSCORE results. However, it should be noted that HABSCORE was developed in England and Wales and its validity in Scotland has not been tested. Consequently, HABSCORE results used within this thesis should be used as indicative only. Ultimately, there is the possibility that abstraction rates and/or the outfall weir may have impacted on 0+ salmon densities at site c in Inverhaggernie Burn in 2011.

A reduction of 0+ trout was also observed in the depleted reaches, post commissioning of hydropower schemes, at Kinnaird Burn, site e in 2010 and 2011, and Inverhaggernie Burn, site b in 2011. Reductions of 0+ individuals were also observed at sites upstream of the intakes, discussed earlier. Again, it is highlighted that 0+ fish spawned/hatched after the hydropower schemes were commissioned can indicate potential effects of abstraction on spawning success. Salmonids spawn in fast-flowing waters (Jonsson & Jonsson, 2011) and the velocity of the water is crucial during the incubation stage. Many of the sites located in the abstracted reach of the schemes under study, support good spawning areas and it is therefore a concern that during hydropower commissioning, the abstracted reach may experience reduced flows, which could hinder egg incubation. During egg incubation, there must be sufficient flow and depth to ensure oxygen is transferred to the eggs and embryos at all times and allow metabolic wastes to be carried away (Gore & Hamilton, 1996). High permeability of the streambed is thus vital within spawning areas to maintain high intra-gravel oxygen concentrations (Cowx & Welcomme, 1998). In the depleted stretches of river, however, the flow and depth of water are likely to be reduced and thus it is unknown whether the remaining flows are sufficient enough to clean the gravels of waste products. The incubation period of salmonid eggs can last for more than 50 days, however this depends primarily on the temperature of the water (Berg & Moen, 1999; Jonsson & Jonsson, 2011); the duration increases with decreasing water temperature. When flows are reduced, however, such as in depleted stretches of river, temperatures can increase due to a reduction in wetted area and increase in shallower water, but also reduction in hyporheic flows, all of which may have consequences for the development and reproduction of aquatic organisms that are influenced by temperature (Floodmark *et al.,* 2004). Additionally, decreases in stream velocity, likely to occur in depleted stretches of river, are thought to reduce food delivery, resulting in a less than optimal habitat for important processes such as Atlantic salmon parr rearing (Bjornn & Peery, 1992).

It is documented that flow reductions affect the physical habitat characteristics of rivers, such as water velocity, sediment transport, turbidity, bed and bank stability (Growns, 2008), wetted width, water depth and water temperature. While much of the changes experienced will depend on the channel morphology, flow modifications such as those induced by run-of-river hydropower schemes have the potential to alter the quantity and quality of available aquatic habitat (Lake, 2003), which subsequently influences stream biota (Anderson *et al.,* 2006). It is therefore possible that the reduction of flow within the depleted reaches of Kinnaird Burn and Inverhaggernie Burn has reduced the spawning habitat available for salmonids by altering the specific environmental conditions that are required. Female trout select their spawning sites based on locally viable criteria of water depth and velocity, availability of nearby cover, gravel size, compaction and porosity and stability (Verspoor *et al.,* 2007). Redds must be created in cold fast flowing waters not only for good oxygen levels but to also mix the sperm and eggs and ensure efficient fertilisation. Accelerated water flow is essential to ensure the gravels contain low concentrations of fine materials such as sand and silt. To fulfil these requirements, spawning tends to be in riffles or faster-flowing areas at the head and tail of pools (Mills, 1989). Females have even been observed to “crouch” in the redd and lower their anal fin in order to test the flow of water in the bottom of the pit (Crisp, 2000). If conditions are unsatisfactory, females will abandon the site and seek out another. Reduced flows in depleted reaches may further expose or render spawning areas too shallow, leading to redd dewatering or freezing of eggs or alevins (Gibbins *et al.,* 2008), compromise hyporheic flows to oxygenate incubating eggs, as well as increase the deposition of fine sediments (Barlaup *et al.,* 2008). These factors have negative consequences for egg survival and development and larval emergence, and could reduce recruitment in salmonid populations. This perhaps was the case at the sites within the depleted reach of Kinnaird Burn and Inverhaggernie Burn resulting in reduced recruitment; indicated by the lack of 0+ individuals in years surveyed post commissioning in the depleted reaches. However, reductions of 0+ trout were also observed at the downstream control sites of Kinnaird Burn (site g) and Inverhaggernie Burn (site d) suggesting changes in 0+ trout densities are likely to be due to natural fluctuations in recruitment.

A reduction in 0+ salmon was also observed at site e, located in the abstracted reach, of Innerhadden Burn between 2010 and 2011. This decline was also noted however, at site f, located downstream of the abstracted reach. As mentioned previously, natural variability in recruitment is a prominent feature of salmonids. Data collected by Marine Scotland[1] identified the number of returning adult salmon (rod catch data) in 2009 and 2010 to the Tay catchment; which Innerhadden Burn is located within. It is generally considered that an increase in the number of returning adults is likely to result in an increase in the number of 0+ fish the subsequent year. Indeed, significant positive relationships have been found between the number of sea winter female salmon and subsequent fry density (Niemela *et al*., 2005). Between 2009 and 2010 there was a slight increase in the number of returning adult salmon (rod catch data) from 2476 (salmon and grilse) to 2901 (salmon and grilse). At site e, however, the densities of 0+ fish caught between 2010 and 2011 decreased from 2/100 m^2 to 1/100 m^2. Generally a positive relationship between the number of adult returns and juvenile fish is expected (Niemela *et al*., 2005). There are many factors, however, discussed above, that can affect the embryo stage of a salmonid’s life cycle. Salmonid eggs are incredibly dependent upon specific environmental conditions and natural fluctuations in temperature or oxygen levels can have extremely negative consequences. For example, embryos spend a substantial period in the hyporheic zone and brief periods of de-oxygenation have been reported as being responsible for lethal/sub-lethal effects which can affect recruitment to populations (Soulsby *et al.,* 2009). Warmer water temperatures and environmental stress such as mechanical disturbances or low oxygen levels, can lead to premature emergence of alevins. As embryonic development proceeds faster at higher temperatures, a natural increase in temperature may lead to premature emergence of larvae, possibly exposing them to spring spate flows, greater predation and scarce food resources, potentially resulting in increased mortality (Saltveit *et al.,* 2001). A combination of these factors could perhaps give reason for the reduced number of 0+ fish in spite of an increase in adult returns.

Increase in density estimates

In some years, densities of salmonids were higher in the abstracted reach post commissioning compared to years pre commissioning. At site d (only), located in the depleted reach of Keltney Burn, 0+ salmon and ≥1+ trout densities were higher in 2011 post commissioning than those observed in previous years pre hydropower commissioning. It should be noted however that these increases were also observed at

[1] Please note that rod catch return figures reported in this thesis are Crown copyright, used with the permission of Marine Scotland Science. Marine Scotland Science is not responsible for interpretation of these data by third parties

site f located outside of the abstracted reach. Similarly at site e (only), also located in the depleted reach ≥1+ trout densities were the highest in 2010 post-commissioning than all years but also higher in 2011 compared to densities observed in 2008 and 2009 pre hydropower commissioning. At Innerhadden Burn, densities of ≥1+ salmon were found to be higher in the abstracted reach at site e in 2011 post hydropower commissioning compared to years prior hydropower commissioning. This was also observed at site f located outside of the abstracted reach. However, it is important to observe that at site f, located outside of the abstracted reach, ≥1+ salmon densities observed in 2011 post abstraction were considerably lower to those observed in 2007 pre abstraction. By contrast, ≥1+ salmon densities in 2011 at site e located in the abstracted reach remained fairly similar to those recorded in 2007. These data suggest that reduced flows have potentially created more favourable spawning (by reducing flows in winter) and rearing (by reducing flows in the summer) habitats, supporting higher densities of fish. This theory however, is not supported by any literature and indeed contradicts other research. The importance of high flows are widely accepted, specifically during the spawning and incubation periods to keep redds free of fine sediment, of which a build up can have negative consequences (Bash *et al.,* 2001; Kemp *et al.,* 2011) (see Chapters 2 & 5) for the eggs and fry. While it could be considered that reduced flows in the depleted stretch have increased survival of eggs and fry, by eliminating the consequences of large spates, other factors must also be taken into account. It is important to note that the natural flow of a river can be tremendously variable between years, differing in duration of low flows and number of spate events. The timing of these events alone and combined with temperature can therefore naturally have significant effects on annual fish recruitment (Lobon-Cervia & Mortensen, 2005; Tetzlaff *et al.,* 2005). It is therefore possible that trends in salmonid densities are in response to natural variability as opposed to the run-of-river hydropower scheme as densities also increased at the control site.

4.8 Conclusion

During the current period of monitoring (i.e. prior to and following commissioning of the hydropower schemes) it is considered there have been possible impacts of flow abstraction from the hydropower schemes on ≥1+ salmon density at Rottal Burn site e and 0+ salmon density at Inverhaggernie Burn site c. No impact of the hydropower schemes on 0+ and ≥1+ trout and salmon populations in Kinnaird Burn, Keltney Burn and Innerhadden Burn were detected. However, while no impact has been detected during this current study at the three latter schemes, based on similar trends in fish density at sites within and outside of the abstracted reach, possibilities of a long-term impact cannot be ruled out. Indeed, salmonid populations in streams may take up to 7 years (or two generations) to respond fully to environmental/habitat changes (Hunt, 1976). Many of the schemes under study have only been operational for 2 years and as such any negative responses to alterations of flow may not be reflected within this short period of time. It should also be acknowledged, that the potential impacts within this study were only analysed on a site by site basis as cumulative effects could not be accounted for within the scope of this study; impacts of the schemes are thus confined to the impacted reach, which indeed may not be the case.

CHAPTER FIVE

5 POTENTIAL IMPACTS OF RUN-OF-RIVER HYDROPOWER SCHEMES ON FISH POPULATIONS: STAGE TWO

5.1 Introduction

This chapter compares the density, age structure and growth of Atlantic salmon and brown trout, before and after hydropower commissioning, as described in Chapter 3 for five of the ten run-of-river hydropower schemes under study. These five schemes are the River Callop, Ardvorlich Burn, Douglas Water, Camserney Burn and Allt Gleann Da-Eig. In comparison to the schemes documented in Chapter 4, these schemes present some limitations regarding analysis and interpretation; three schemes present issues relating to the monitoring design, one has post operational data only and one had not become operational until after the study period, thus analysis was restricted to the detection of impacts during the construction phase. While conclusions are therefore surrounded with an element of caution in some cases, analysis of data aims to isolate natural trends that are observed in fish populations from trends potentially in response to operation of the hydropower schemes.

5.2 River Callop hydropower scheme

5.2.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes. The EA-FCS assessments were not possible at sites surveyed by MFC in 2006 or LFT in 2010 as individual run data were not available to allow calculations. The data supplied by LFT could only be analysed using the SFCC national and regional classification schemes as no river width data were available and the EA-FCS is based on triple run or calibrated single run data.

It should be noted that surveys carried out by HIFI (the author carried out these surveys as part of the HIFI team) and LFT were at sites surveyed by MFC in 2006 for comparative purposes; the latter surveys involved no assessment of fish populations upstream of the intakes therefore no comparative control sites in the upper reaches exist. Sites c5, c3, c6, c7, c2 and c4 were all located within the abstracted reach with only sites cH and c1 outside the abstracted reach and located downstream of the outfall.

Depleted reach

No salmon were caught in the upper reaches of the River Callop throughout the study period. No trout were caught or observed at site c5, downstream of intake 3 on Allt na Cruaiche, throughout the study period (Figure 5.1, Table 5.1). At site c3, downstream of the intake on Allt an Fhaing, 0+ trout populations decreased from fair/average (class C) in 2006 to zero in 2011 (Table 5.1). ≥1+ trout populations at site c3 increased from 2006 to 2010 but were at the lowest level in 2011 (Figure 5.1); populations were fair/average (class C) in 2006 and 2010 and fair/poor (class D) in 2011 (Table 5.1).

Table 5.1 Classification of trout densities in both the upper and lower reaches of the River Callop between 2006 and 2011 based on five classification schemes. Note hydropower scheme commissioned in 2008. In 2006, sites c1, c2 and c4 were triple run but EA-FCS could not be calculated as information on each run was unavailable. Classifications could not be produced for the different classification schemes regarding the 2010 LFT densities as river width information was unavailable. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c5*					
2006	F	F	F	F	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	F	F	F	F	F
2011	F	F	F	F	F
c3*					
2006	C	C	D	E	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	E	E	E	E	E
2011	F	F	F	F	F
c6*					
2006	D	D	D	D	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	C	D	E	C
2011	E	E	E	E	E
c7*					
2006	E	D	E	C	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	C	C	C	C
2011	E	E	E	E	E
c2*					
2006	B	C	C	D	
2010 L	C	C			
2010 H	C	C	C	C	C
2011	D	D	D	D	D
c4*					
2006	E	D	D	C	
2010 L	C	C			
2010 H	N/S	N/S	N/S	N/S	N/S
2011	N/S	N/S	N/S	N/S	N/S
cH					
2006	n/s	n/s	n/s	n/s	n/s
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	C	B	A	C
2011	D	D	B	B	D
c1					
2006	C	C	B	A	
2010 L	C	C			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	E	E	E	E	E

≥1+ trout

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c5*					
2006	F	F	F	F	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	F	F	F	F	F
2011	F	F	F	F	F
c3*					
2006	C	C	E	E	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	C	E	D	C
2011	D	D	E	E	E
c6*					
2006	F	F	F	F	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	D	D	E	E	D
2011	E	E	E	E	E
c7*					
2006	E	E	E	E	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	E	E	E	E	E
2011	E	E	E	E	E
c2*					
2006	F	F	F	F	
2010 L	E	E			
2010 H	B	B	B	B	C
2011	E	E	E	E	E
c4*					
2006	E	E	E	E	
2010 L	E	E			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	n/s	n/s	n/s	n/s	n/s
cH					
2006	n/s	n/s	n/s	n/s	n/s
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	E	E	C	C	E
2011	F	F	F	F	E
c1					
2006	F	F	F	F	
2010 L	F	F			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	E	E	E	E	E

At site c6, in the lower reaches of Allt Tarsuinn and downstream of intakes 1 and 2, 0+ trout populations increased from 2006 to 2010 but were at the lowest level in 2011 (Figure 5.1); populations were fair/poor (class D) in 2006, fair/average (class C) in 2010 and poor (class E) in 2011 (Table 5.1). ≥1+ trout were only captured at site c6 in 2010 and 2011 and populations were fair/poor (class D) and poor (class E) respectively (Figure 5.1, Table 5.1). 0+ salmon were only caught at site c6 in 2006 and populations were good (class B). ≥1+ salmon populations were highest in 2010 and lowest in 2011 ranging from fair/average (class C) in 2006 and 2010 to poor (class E) in 2011 (Figure 5.1,Table 5.2).

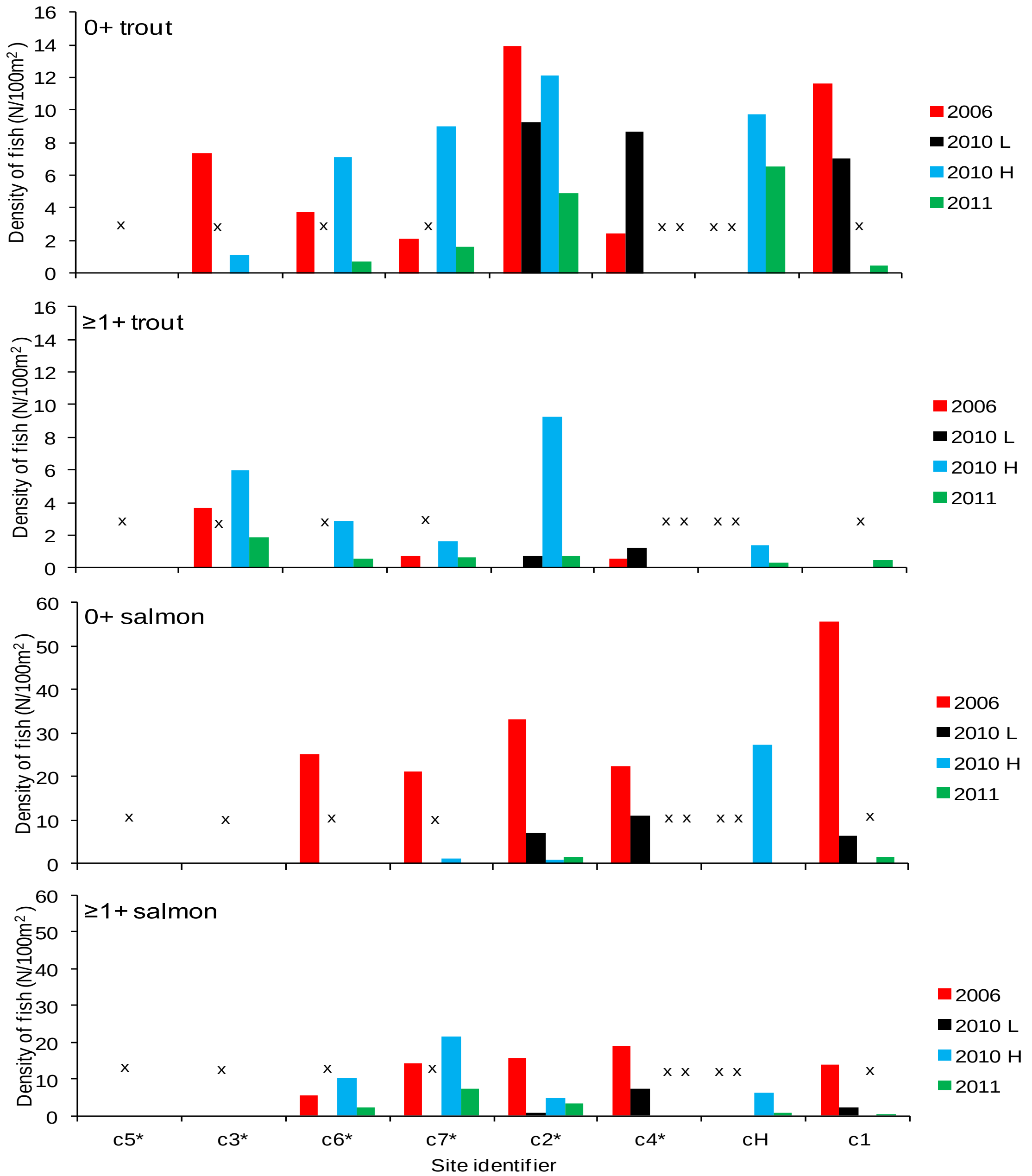

Figure 5.1 Density estimates of trout and salmon in the River Callop between 2006 and 2011. Note in 2010 there were two density estimates at site c2 as both LFT and HIFI surveyed the site. L = LFT, H = HIFI. Hydropower scheme was commissioned in 2008. x represents years not surveyed.

Table 5.2 Classification of salmon densities in the upper and lower reaches of the River Callop between 2006 and 2011 based on five classification schemes. In 2006, sites c1, c2 and c4 were triple run but EA-FCS could not be calculated as information on each run was unavailable. Classifications could not be produced for the different classification schemes regarding the 2010 LFT densities as river width information was unavailable. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c6*					
2006	B	B	C	B	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	F	F	F	F	F
2011	F	F	F	F	F
c7*					
2006	B	B	C	A	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	E	E	E	E	E
2011	F	F	F	F	F
c2*					
2006	B	A	B	A	
2010 L	D	C			
2010 H	E	E	E	E	E
2011	E	E	E	E	E
c4*					
2006	B	B	B	A	
2010 L	C	C			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	n/s	n/s	n/s	n/s	n/s
cH					
2006	n/s	n/s	n/s	n/s	n/s
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	C	C	C	C
2011	F	F	F	F	F
c1					
2006	A	A	A	A	
2010 L	D	C			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	E	E	E	E	E

≥1+ salmon

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c6*					
2006	C	C	C	C	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	C	B	C	B	B
2011	E	D	E	D	D
c7*					
2006	B	A	B	A	
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	B	A	B	A	A
2011	C	B	C	B	B
c2*					
2006	B	A	B	A	
2010 L	E	E			
2010 H	D	D	D	D	C
2011	E	E	E	E	D
c4*					
2006	A	A	A	A	
2010 L	C	B			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	n/s	n/s	n/s	n/s	n/s
cH					
2006	n/s	n/s	n/s	n/s	n/s
2010 L	n/s	n/s	n/s	n/s	n/s
2010 H	E	E	E	D	C
2011	E	E	E	E	E
c1					
2006	B	B	B	A	
2010 L	E	D			
2010 H	n/s	n/s	n/s	n/s	n/s
2011	E	E	E	E	E

At site c7, in the lower reaches of Allt na Cruaiche and immediately upstream of the confluence with the River Callop, 0+ trout populations increased from 2006 to 2010 but were at the lowest level in 2011 but similar to densities in 2006 (Figure 5.1); populations were fair/average (class C) in 2010 and poor (class E) in 2006 and 2011 (Table 5.1). ≥1+ trout populations were low at site c6 throughout the study period and classified as poor (class E). 0+ salmon populations at site c7 decreased over the study period from good (class B) in 2006 to absent (class F) in 2011. ≥1+ salmon populations were highest in 2010 and lowest in 2011 ranging from good (class B) in 2006 and 2010 to fair/average (class C) in 2011 (Figure 5.1,Table 5.2).

At site c2, in the lower reaches of Allt an Fhaing and upstream of the confluence with the River Callop, 0+ trout populations decreased from 2006 to 2011 and were good (class B) in 2006 decreasing to fair/poor (class D) in 2011 (Table 5.1); surveys carried out by LFT and HIFI in 2010 differed in density estimates but both classified populations as fair/average (class C). ≥1+ trout populations were generally low, except in 2010, and classified as poor (class E) in 2011 and absent (class F) in 2006 but good (class B) in 2010 (Table 5.1); surveys carried out by LFT and HIFI in 2010 differed markedly in density estimates and populations were classified as poor (class E) and good (class B) respectively. 0+ salmon populations at site c2 decreased over the study period from good (class B) in 2006 to poor (class E) in 2011. Surveys carried out by LFT and HIFI in 2010 differed in 0+ salmon density estimates and populations were classified as fair/poor (class D) and poor (class E) respectively. ≥1+ salmon populations were highest in 2006 and lowest in 2011, but surveys by LFT recorded lowest densities in 2010; populations ranged from good (class B) in to poor (class E) in 2011 (Figure 5.1, Table 5.2).

At site c4, in the lower reaches of Allt na-Cruaiche, surveys were only carried out in 2006 (MFC) and 2010 (LFT). 0+ trout populations increased from 2006 to 2010 and were poor (class E) in 2006 increasing to fair/average (class C) in 2010 (Figure 5.1, Table 5.1). ≥1+ trout populations were low and classified as poor (class E) in 2006 and 2010. 0+ salmon populations decreased from 2006 to 2010 and were good (class B) in 2006 decreasing to fair/average (class C) in 2010 (Figure 5.1, Table 5.2). ≥1+ salmon populations also decreased from 2006 to 2010 and were excellent (class A) and fair/average (class C) respectively.

Downstream of outfall

Site cH, downstream of the outfall, was not surveyed prior to hydropower commissioning but was surveyed by HIFI in 2010 and 2011 as an additional control if data collected at site c1 in 2010, the control site surveyed in 2006, were not available from LFT; however these data were subsequently kindly provided by LFT. Site cH was included in the analysis to identify any trends in salmonid populations immediately downstream of the outfall. 0+ trout populations at site cH decreased from 2010 to 2011, as found at sites within the abstracted reach, and were fair/average (class C) in 2010 and fair/poor (class D) in 2011 (Figure 5.1, Table 5.1). ≥1+ trout populations were low at cH, as found at sites within the abstracted reach, and classified as poor (class E) in 2010 and 2011 (Figure 5.1, Table 5.1). 0+ salmon were only caught at site cH in 2010 and populations were fair/average (class C) with no deviation in classification using the various SFCC schemes (Figure 5.1, Table 5.2). ≥1+ salmon populations decreased from 2010 to 2011 and populations were poor (class E) in both years.

At site c1, downstream of the outfall, 0+ trout populations decreased from 2006 to 2011 and were fair/average (class C) in 2006 and 2010 and poor (class E) in 2011 (Figure 5.1, Table 5.1). ≥1+ trout populations were zero in 2006 and 2011 and poor (class E) in 2011). 0+ salmon populations at site c1 decreased markedly from 2006 to 2011 and were excellent (class A) in 2006 decreasing to poor (class E) in 2011. ≥1+ salmon populations also decreased from 2006 to 2011 and were good (class B) in 2006 decreasing to poor (class E) in 2011 (Figure 5.1, Table 5.2).

5.2.2 Salmonid size structure

Individual salmonid length data were not available from surveys carried out in 2006 or from surveys carried out by LFT in 2010, therefore, it was not possible to identify if there were any changes in size structure pre and post hydropower commissioning. However, an overview of the size structure, from surveys in 2010 and 2011 by HIFI, was provided to give an indication of size structure and to attempt to identify if there had been any changes between 2010 and 2011.

Depleted reach

At site c3, downstream of the intake on Allt an Fhaing, only one 0+ trout was caught in 2010 and none were present in 2011; ≥1+ trout were caught in a narrow size range in both years. At site c6, 0+ salmon were absent in 2010 and 2011 while ≥1+ salmon were caught in the size ranges 85-125 mm and 75-95 mm in 2010 and 2011 respectively. 0+ trout were caught in the size range 45-75 mm in 2010 while only one 0+ trout was caught in 2011; only two ≥1+ trout and one ≥1+ trout were caught in 2010 and 2011 respectively.

At site c7, 0+ salmon were absent in 2010 and only one 0+ salmon was caught in 2011; ≥1+ salmon were caught in the size ranges 75-130 mm and 75-105 mm in 2010 and 2011 respectively (Figure 5.2). 0+ trout were caught in the size range 45-65 mm in 2010 while only one 0+ trout was caught in 2011; only two ≥1+ trout and one ≥1+ trout were caught in 2010 and 2011 respectively (Figure 5.3).

At site c2, only one 0+ salmon was caught in 2010 while salmon were caught in the size range 40-50 mm in 2011; ≥1+ salmon were caught in the size ranges 85-145 mm and 70-100 mm in 2010 and 2011 respectively (Figure 5.4). 0+ trout were caught in the size range 45-65 mm in 2010 and 40-65 mm in 2011; ≥1+ trout were caught in the size range 80-160 mm in 2010 while only one ≥1+ trout was caught in 2011 (Figure 5.5).

Downstream of outfall

At site cH 0+ salmon in 2010 were caught in the size range 35-60 mm but were absent in 2011; ≥1+ salmon were caught in the size range 80-200 mm in 2010 and a narrower size range of 85-95 mm in 2011 (Figure 5.6). 0+ trout were caught in the size range 40-70 mm and 45-65 mm in 2010 and 2011 respectively; ≥1+ trout were caught in the size range 85-120 mm in 2010 while only one ≥1+ trout was caught in 2011 (Figure 5.7).

Length data were only available at site c1 in 2011 with only ≥1+ salmon caught, in the size range 75-85 mm, and only one 0+ trout and one ≥1+ trout caught in 2011.

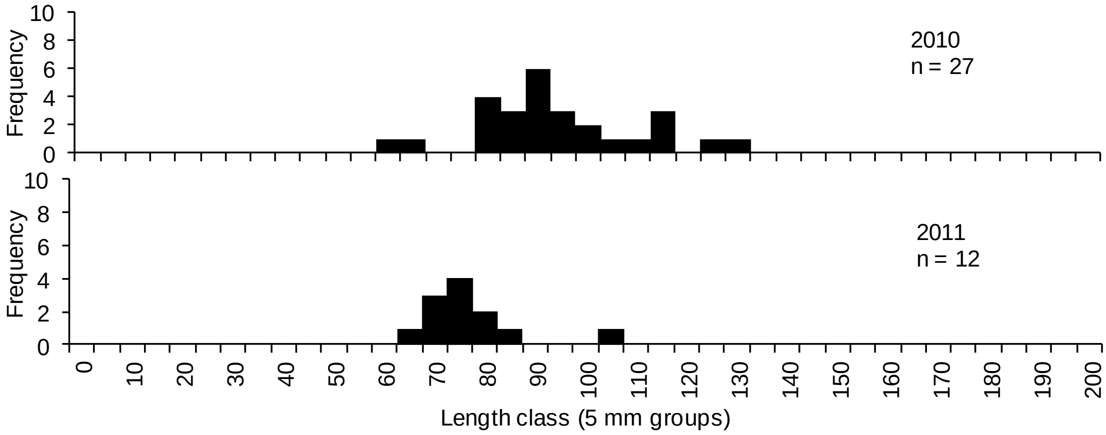

Figure 5.2 Length distributions of salmon at site c7 in the River Callop in 2010 and 2011.

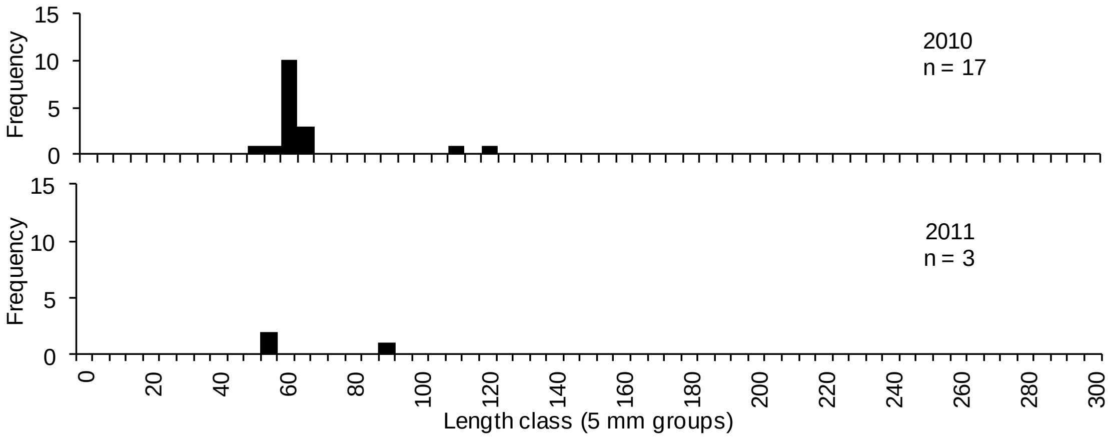

Figure 5.3 Length distributions of trout at site c7 in the River Callop in 2010 and 2011.

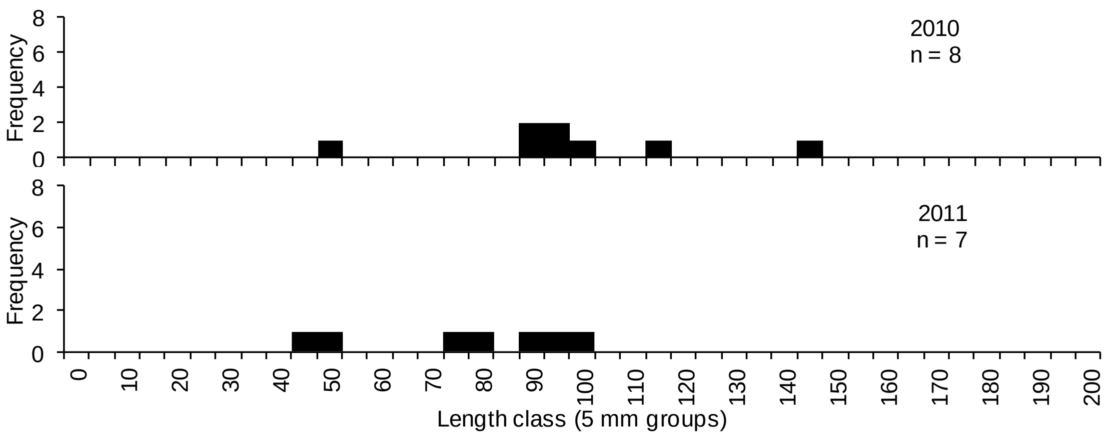

Figure 5.4 Length distributions of salmon at site c2 in the River Callop in 2010 and 2011.

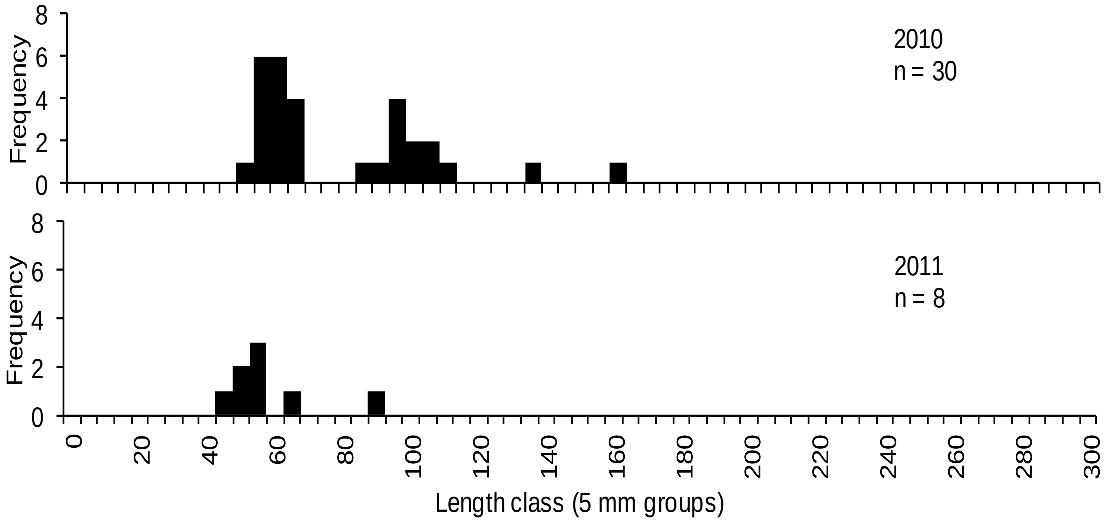

Figure 5.5 Length distributions of trout at site c2 in the River Callop in 2010 and 2011.

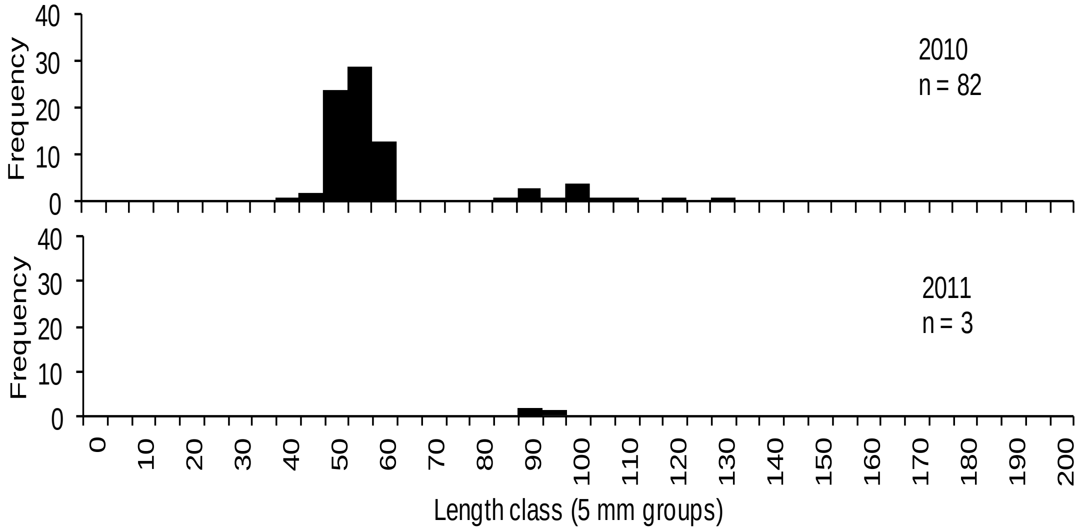

Figure 5.6 Length distributions of salmon at site cH in the River Callop in 2010 and 2011.

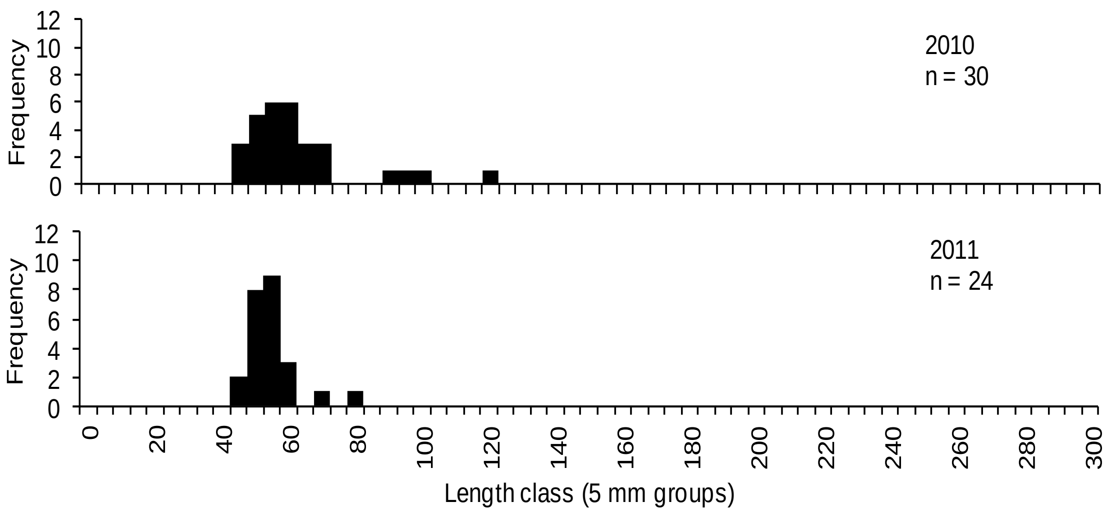

Figure 5.7 Length distributions of trout at site cH in the River Callop in 2010 and 2011.

5.2.3 HABSCORE analysis

HABSCORE data were not collected in surveys in 2006 by MFC but data were collected by HIFI in 2010 and 2011. Although the data will not allow any assessment of potential impact of the hydropower scheme the analysis is included to identify the potential of each site for salmonid usage in 2010 and 2011 (Table 5.3).

The observed densities of 0+ salmon at all sites were lower than predicted by the Habitat Quality Score (HQS) suggesting poorer populations than expected; the Habitat Utilisation Index (HUI) upper CLs were all <1 therefore the observed populations were significantly lower than expected (Table 5.3). The observed densities of ≥1+ salmon at sites c2, c7, c6, and cH were higher than predicted (HQS) suggesting better populations than expected, the HUI lower CL was >1 at site c7 therefore the observed population was significantly higher than expected (Table 5.3). The observed density of ≥1+ salmon at site c1 was lower than predicted (HQS), suggesting poorer populations than expected but the upper HUI CL was >1 therefore the observed population was not significantly lower than expected (Table 5.3).

The observed densities of 0+ trout at all sites were lower than predicted by the HQS suggesting poorer populations than expected; the HUI upper CLs were <1 at sites c6, c5, c3 and c1 therefore the observed populations were significantly lower than expected at these sites (Table 5.3).

The observed densities of 0+ trout (<20 cm) at all sites were lower than predicted by the HQS suggesting poorer populations than expected; the HUI upper CLs were <1 at sites c7 and c6 therefore the observed populations were significantly lower than expected at these sites (Table 5.3).

The observed densities of 0+ trout (>20 cm) at all sites were zero and lower than predicted by the HQS suggesting poorer populations than expected but the HUI upper CLs were >1 therefore the observed populations were not significantly lower than expected (Table 5.3).

Table 5.3 HABSCORE outputs for survey sites after (2005-2011) the River Callop was commissioned. Note: shaded area represents sites where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions). Salmon populations were not analysed at sites c3 and c5 as they are upstream of impassable barriers.

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2010-2011 c2	1.4	1.0	19.86	5.74	68.69	0.05	0.01	0.37	-3.00
2010-2011 c7	1.4	0.88	12.05	3.74	38.84	0.07	0.01	0.52	-2.66
2010-2011 c6	0	0	45.64	12.20	170.80	0.01	0.00	0.10	-4.61
2010-2011 c5	0	0							
2010-2011 c3	0	0							
2010-2011 cH	8.8	2.36	175.59	42.10	732.31	0.01	0.00	0.11	-4.61
2010-2011 c1	3.0	1.42	184.56	45.66	746.04	0.01	0.00	0.06	-4.61
≥1+ salmon									
2010-2011 c2	5.9	4.18	1.25	0.34	4.60	3.35	0.50	22.36	1.21
2010-2011 c7	29.4	18.33	2.33	0.66	8.23	7.87	1.22	50.80	2.06
2010-2011 c6	8.9	4.97	1.77	0.51	6.15	2.82	0.44	17.99	1.04
2010-2011 c5	0	0							
2010-2011 c3	0	0							
2010-2011 cH	6.9	1.86	1.36	0.37	4.90	1.37	0.21	9.03	0.31
2010-2011 c1	1.0	0.47	1.51	0.42	5.43	0.31	0.05	2.03	-1.17
0+ trout									
2010-2011 c2	10.9	7.70	29.20	7.38	115.58	0.26	0.04	1.82	-1.35
2010-2011 c7	8.1	5.06	30.42	7.67	120.67	0.17	0.02	1.15	-1.77
2010-2011 c6	3.2	1.76	20.91	5.38	81.25	0.08	0.07	0.57	-2.53
2010-2011 c5	0	0	6.32	1.62	24.62	0.09	0.01	0.59	-2.41
2010-2011 c3	1.0	0.90	14.74	3.77	57.71	0.06	0.01	0.42	-2.81

Table 5.3 continued.

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ trout									
2010-2011 cH	24.9	6.69	7.10	1.82	27.73	0.94	0.14	5.43	-0.06
2010-2011 c1	1.0	0.47	13.50	3.40	53.67	0.03	0.01	0.24	-3.51
≥1+ trout (<20 cm)									
2010-2011 c2	3.6	2.55	7.36	1.65	32.88	0.35	0.06	2.15	-1.05
2010-2011 c7	1.7	1.08	8.44	1.82	39.17	0.13	0.02	0.82	-2.04
2010-2011 c6	1.7	0.96	10.38	2.36	45.70	0.09	0.02	0.57	-2.41
2010-2011 c5	0	0	2.70	0.57	12.70	0.20	0.03	1.31	-1.61
2010-2011 c3	3.5	3.13	6.04	1.34	27.25	0.52	0.08	3.2	-0.65
2010-2011 cH	2.0	0.54	0.71	0.15	3.27	0.76	0.11	5.07	-0.27
2010-2011 c1	1.0	0.47	1.33	0.30	5.91	0.35	0.06	2.19	-1.05
≥1+ trout (>20 cm)									
2010-2011 c2	0	0	0.37	0.12	0.17	1.92	0.60	6.11	0.65
2010-2011 c7	0	0	0.27	0.09	0.82	2.34	0.76	7.16	0.85
2010-2011 c6	0	0	1.09	0.34	3.48	0.51	0.16	1.64	-0.67
2010-2011 c5	0	0	0.50	0.18	1.80	0.95	0.30	7.99	-0.05
2010-2011 c3	0	0	1.77	0.55	5.77	0.51	0.16	1.66	-0.67
2010-2011 cH	0	0	0.09	0.03	0.30	2.89	0.89	9.35	1.06
2010-2011 c1	0	0	0.30	0.10	0.93	1.58	0.51	4.93	0.46

5.2.4 Salmonid growth rates

Growth rate information was not available from surveys carried out by MFC in 2006 or LFT in 2010; data were only available for fish captured in surveys by HIFI in 2010 and 2011 limiting the identification of any growth trends over time.

Depleted reach

At site c3 back-calculated length of trout at age 1 was 75 mm and age 2 was 113 mm in 2009. At site c6, back-calculated length of trout at age 1 was 55 mm in 2008 and 2010 and 51 mm in 2011; back calculated length of trout at age 2 was 109 mm in 2008 (Table 5.4). At site c7, back-calculated length of trout at age 1 ranged between 59 and 72 mm, with the slowest growth in 2010 and fastest in 2009, excluding 2011 data as these were based on observed mean length (Table 5.4). At site c2, back-calculated length of trout at age 1 ranged between 57 and 63 mm, excluding 2011 data, with the slowest growth in 2010 and fastest in 2009; back-calculated length of trout at age 2 was 124 mm in 2008 (Table 5.4).

Table 5.4 Back-calculated lengths at age of trout in the upper and lower reaches of the River Callop. Note no 0+ trout were captured at site c3 in 2011. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))	
		1	2
c3	2009	75±10 (3)	113(1)
c6	2008	55(1)	109(1)
	2010	55(1)	
	*2011	51(1)	
c7	2009	72±11(2)	
	2010	59(1)	
	*2011	53±1(2)	
c2	2008	62(1)	124(1)
	2009	63±7(9)	
	2010	57(1)	
	*2011	51±6(7)	
cH	2010	53(1)	
	*2011	52±6(23)	
c1	2010	58(1)	
	*2011	54(1)	

At site c6, back-calculated length of salmon at age 1 ranged between 54 and 64 mm, with the slowest growth in 2009 and fastest in 2010; a similar trend in slow and fast growth was found at site c7 (Table 5.5). At site c2, back-calculated length of salmon at age 1 ranged between 54 and 62 mm, excluding 2011 data, with the slowest growth in 2010 and fastest in 2009. At site c4, back-calculated length of salmon at age 1 ranged between 53 and 56 mm, with the slowest growth in 2008 and fastest in 2009; back calculated length of salmon at age 2 was 88 mm in 2008 (Table 5.5).

Downstream of the outfall

At site cH, back-calculated length of trout at age 1 was 53 mm in 2010 (Table 5.4), while at site c1, back-calculated length of trout at age 1 was 58 mm in 2010. Back-calculated length of salmon at age 1 was 56 mm in 2010 while at site c1 back-calculated length of salmon at age 1 was 51 mm in 2010 (Table 5.5).

Table 5.5 Back-calculated lengths at age of salmon in the lower reaches of the River Callop. Note no 0+ salmon were caught in 2011 at sites c7, cH, and c6. Site c4 was not sampled in 2011. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))	
		1	2
c6	2009	64±8(7)	
	2010	54±6(4)	
c7	2009	58±9(20)	
	2010	48±5(8)	
c2	2009	62±9(5)	
	2010	54±5(5)	
	*2011	44±4(2)	
c4	2008	53(1)	88(1)
	2009	56±4(11)	
cH	2010	56±6(3)	
c1	2010	51(1)	
	*2011	77±2(3)	

5.2.5 Eel

In 2006 eels were caught at sites c7, c1, c2, c3, c4 and c5. Numbers of eels caught at each of the sites were recorded by MFC in the range 1-10, therefore length data and exact abundance at each of the sites were not available. Numbers of eels captured in the River Callop varied between survey years and data suggest the upper and lower reaches of the River Callop are used by small numbers of eels (Table 5.6). Despite slight variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers with the hydropower scheme operational.

Table 5.6 Number of eel captured at survey sites by HIFI in the lower reaches of the River Callop in 2010 and 2011. *Note site c1 was sampled in 2010 by the LFT but eel data were not recorded.

Year	Site identifier					
	c3	c6	c7	c2	cH	c1
2010	3	0	6	13	5	*
2011	1	1	1	5	4	1

5.3 Ardvorlich Burn hydropower scheme

5.3.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes. It should be noted that surveys carried out by SEPA in 2011 were at sites surveyed by MFC in 2008 for comparative purposes; surveys involved no assessment of fish populations upstream of the intakes on the east tributary therefore no comparative control sites in this reach exists, and the control site upstream of the intake on the west tributary will be used. The EA-FCS assessment was only possible at site A1 surveyed by SEPA in 2011 as individual run data were not available to allow calculations at other sites. No salmon were caught in the upper reaches, sites A6, A5 and A4, of Ardvorlich Burn throughout the study period.

Upstream of the intakes

Trout were absent from site A6 throughout the study period (Table 5.7, Figure 5.8).

Table 5.7 Classification of trout densities in both the upper and lower reaches of Ardvorlich Burn in 2008 and 2011 based on five classification schemes. Note hydropower scheme commissioned in 2011. Site A1 and A2 were triple runs in 2008 but trout caught were not split into individual runs, therefore EA-FCS could not be calculated. In 2011 only site A1 was triple run. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
A6					
2008	F	F	F	F	
2011	F	F	F	F	
A5*					
2008	E	E	E	E	
2011	F	F	F	F	
A4*					
2008	F	F	F	F	
2011	E	E	E	E	
A3*					
2008	E	E	E	E	
2011	E	E	E	E	
A2					
2008	D	E	C	D	
2011	C	D	C	D	
A1					
2008	B	C	B	C	
2011	B	C	B	C	B

≥1+ trout

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
A6					
2008	F	F	F	F	
2011	F	F	F	F	
A5*					
2008	C	C	C	D	
2011	A	A	A	A	
A4*					
2008	F	F	F	F	
2011	B	B	B	B	
A3*					
2008	C	C	C	D	
2011	B	C	B	C	
A2					
2008	A	A	A	A	
2011	A	A	A	A	
A1					
2008	C	C	C	D	
2011	A	A	A	A	B

Depleted reach

At sites A5 and A4, in the abstracted reach, 0+ trout densities were low or zero during the study period. 0+ trout populations at site A5 decreased from poor (class E) in 2008 to absent (class F) in 2011 while at site A4 0+ trout populations increased from absent (class F) to poor (class E) (Table 5.7). ≥1+ trout populations at sites A5 and A4 increased between 2008 and 2011, from fair/average (class C) to excellent (class A) at site A5 and from absent (class F) to good (class B) at site A4 (Table 5.7). At site A3 in the lower reaches of Ardvorlich Burn, upstream of the outfall, salmon were absent and 0+ trout populations were similar and poor (class E) in 2008 and 2011 (Figure 5.8, Table 5.7). ≥1+ trout populations at site A3 increased from fair/average (class C) in 2008 to good (class B) in 2011 (Table 5.7).

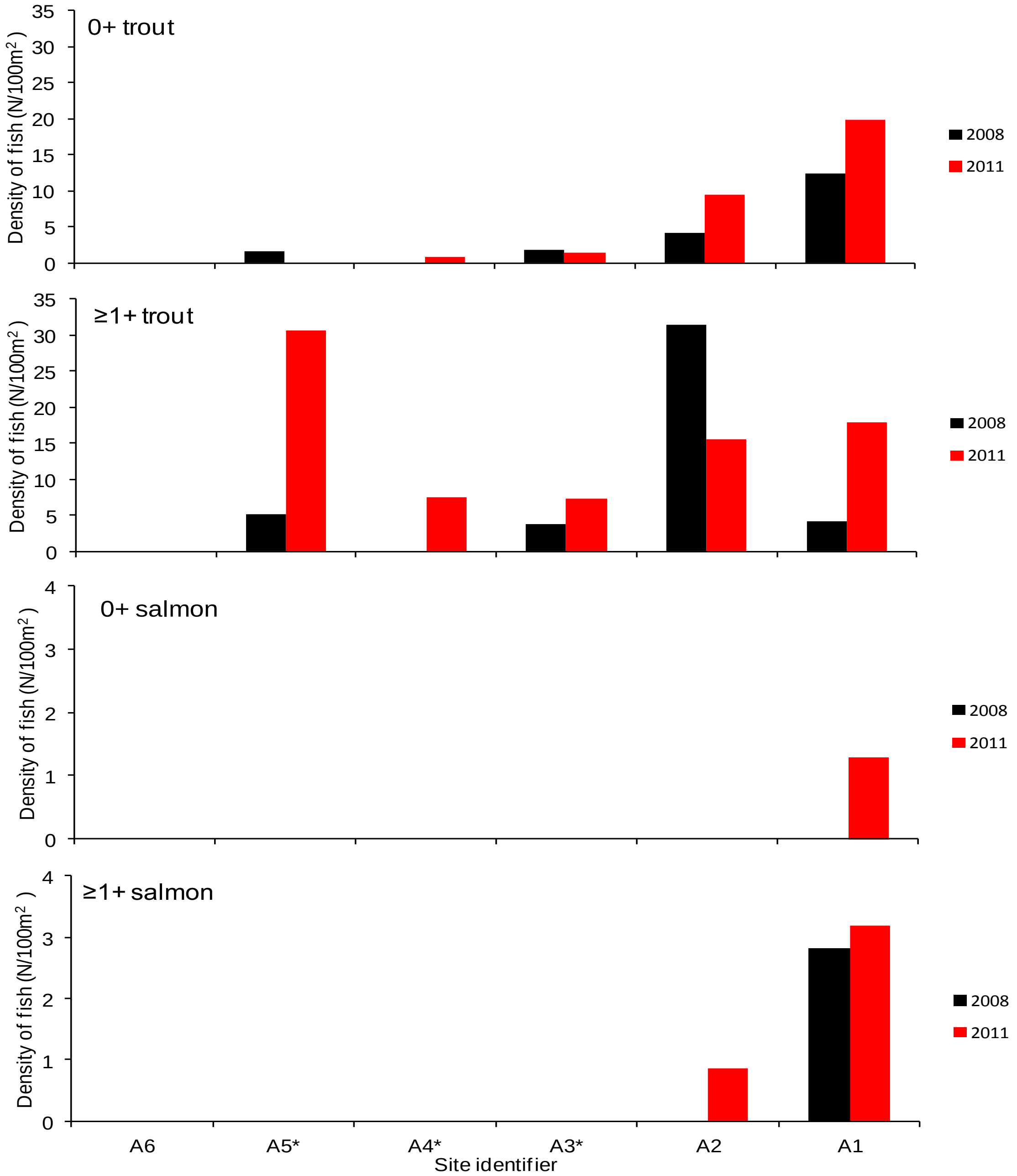

Figure 5.8 Density estimates of trout and salmon in Ardvorlich Burn in 2008 and 2011. Hydropower scheme commissioned in 2011.

Downstream of the outfall

At site A2 downstream of the outfall, 0+ trout populations increased between 2008 and 2011 and were fair/poor (class D) in 2008 and fair/average (class C) in 2011 (Table 5.7). ≥1+ trout populations decreased between 2008 and 2011 but were excellent (class A) in both years (Figure 5.8, Table 5.7). 0+ salmon were absent from site A2 in both study years and ≥1+ salmon were only present in 2011 when populations were poor (class E) (Figure 5.8, Table 5.8). At site A1 further downstream of site A2, 0+ trout populations increased between 2008 and 2011 and were good (class B) in both years (Table 5.7). ≥1+ trout populations also increased between 2008 and 2011 increasing from fair/average (class C) in 2008 to excellent (class A) in 2011 (Figure 5.8, Table 5.7). 0+ salmon were absent from site A1 in 2008 and populations were poor (class E) in 2011 (Table 5.8). ≥1+ salmon populations increased between 2008 and 2011 and were fair/poor (class D) in both years.

Table 5.8 Classification of salmon densities in the lower reaches of Ardvorlich Burn in 2008 and 2011, based on five classification schemes. Note hydropower scheme commissioned in 2011. Site A1 and A2 were triple runs in 2008 but trout caught were not split into individual runs, therefore EA-FCS could not be calculated. In 2011 only site A1 was triple run. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
A2					
2008	F	F	F	F	
2011	F	F	F	F	
A1					
2008	F	F	F	F	
2011	E	E	E	E	E

≥1+ salmon

Site/Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
A2					
2008	F	F	F	F	
2011	E	E	E	E	
A1					
2008	D	E	E	E	
2011	D	D	D	D	D

5.3.2 Salmonid size structure

Individual salmonid length data were not available from surveys carried out in 2008, therefore, it was not possible to identify if there were any changes in size structure pre and post hydropower commissioning. However, an overview of the size structure, from surveys in 2011 by SEPA is provided to give an indication of size structure.

Depleted reach

At site A5, in the abstracted reach, 0+ trout were absent in 2011 and ≥1+ trout were caught in the size range 75-215 mm (Figure 5.9).

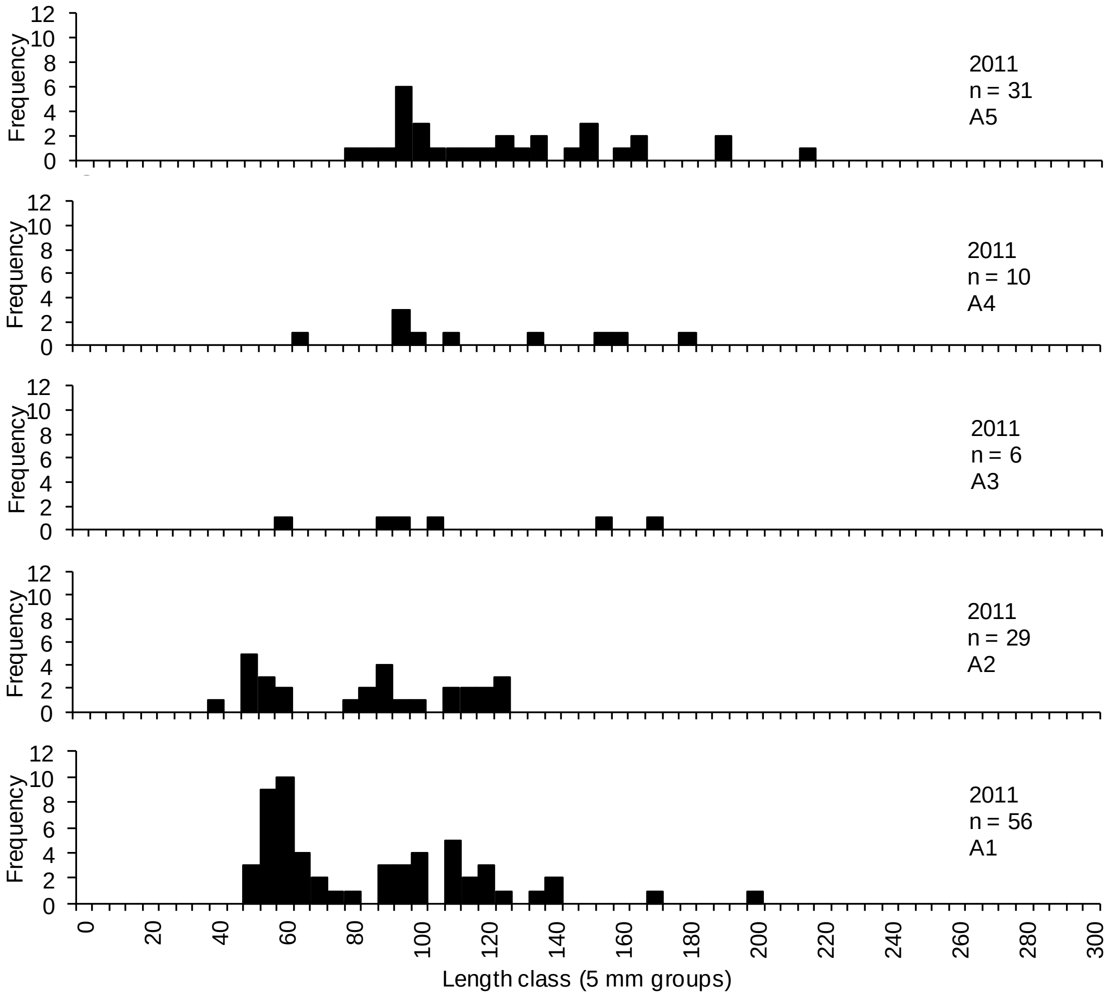

Figure 5.9 Length distributions of trout in the upper and lower reaches of Ardvorlich Burn in 2011.

At site A4, in the abstracted reach, only one 0+ trout was caught in 2011 and ≥1+ trout were caught in the size range 85-175 mm (Figure 5.9). At site A3 in the lower reaches of Ardvorlich Burn and upstream of the outfall, only one 0+ trout was caught in 2011 and ≥1+ trout were caught in the size range 85-170 mm (Figure 5.9).

Downstream of the outfall

At site A2, 0+ trout and ≥1+ trout in 2011 were caught in the size ranges 35-60 mm and 75-125 mm respectively. At site A1, 0+ trout and ≥1+ trout in 2011 were caught in the size ranges 45-70 mm and 75-200 mm respectively (Figure 5.9). Only one salmon (≥1+ individual) was caught at site A2 in 2011 while at site A1 0+ salmon and ≥1+ salmon were caught in the size ranges 50-65 mm and 95-135 mm respectively.

5.3.3 Eel

Eel were recorded at site A1 in 2008,; abundance and length data were unfortunately not recorded by MFC in 2008. Three eels were captured at site A1 in 2011 measured at 263, 380 and 420mm.

5.4 Douglas water hydropower scheme

5.4.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

It should be noted that surveys carried out by SEPA in 2011 were at sites surveyed by MFC in 2002 for comparative purposes; surveys involved no assessment of fish populations upstream of the intake on the south tributary therefore no comparative control sites in this reach exists, and the control site upstream of the intake on Douglas Water was used. The EA-FCS assessment was only possible at site D6 surveyed by SEPA in 2011 as individual run data were not available to allow calculations at other sites; the EA-FCS is based on triple run data or single run calibrated.

No salmon were caught in the any of the survey reaches in Douglas Water.

Upstream of the intake

At site D8, upstream of the intake on Douglas Water, 0+ trout populations decreased between 2002 and 2011 and were fair/average (class C) and poor (class D) respectively (Figure 5.10,Table 5.9). ≥1+ trout populations at site D8 were similar in 2002 and 2011 and were fair/poor (class D) (Table 5.9).

Depleted reach

At site D7, downstream of the intake on the south tributary, 0+ and ≥1+ trout populations were higher in 2011 than in 2002. 0+ trout populations were fair/poor (class D) in both years and ≥1+ trout populations increased from fair/poor (class D) in 2002 to fair/average (class C) in 2011.

At site D6, downstream of the confluence of Douglas Water and Allt Nam Muc and within the abstracted reach, 0+ and ≥1+ trout populations were lower in 2011 than 2002 as found at site D3 downstream of the outfall (Figure 5.10). 0+ trout populations at site D6 decreased from fair/average (class C) in 2002 to fair/poor (class D) in 2011 while ≥1+ trout populations decreased from good (class B) to fair/average (class C) (Table 5.9).

At site D5, upstream of the outfall, 0+ and ≥1+ trout populations were higher in 2011 than 2002, with 0+ populations improving from poor (class E) in 2002 to fair/poor (class D) in 2011; ≥1+ trout populations were fair/average (class C) in both years (Figure 5.10,Table 5.9).

Downstream of the outfall

At site D3, downstream of the outfall, 0+ and ≥1+ trout populations were lower in 2011 than 2002 (Figure 5.10). 0+ trout populations decreased from fair/poor (class D) in 2002 to poor (class E) in 2011 while ≥1+ trout populations decreased from excellent (class A) to good (class B) (Table 5.9).

Table 5.9 Classification of trout densities in both the upper and lower reaches of Douglas Water in 2002 and 2011 based on five classification schemes. Note hydropower scheme commissioned 2008. All sites were triple runs in 2002 but trout caught were not split into individual runs, therefore EA-FCS could not be calculated. In 2011 only site D6 was triple run. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
D8						
	2002	C	C	B	A	
	2011	D	D	B	B	
D7*						
	2002	D	D	C	B	
	2011	D	D	E	E	
D6*						
	2002	C	C	D	D	
	2011	D	D	C	C	E
D5*						
	2002	E	E	C	C	
	2011	D	D	B	B	
D3						
	2002	D	D	D	B	
	2011	E	E	D	D	

≥1+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
D8						
	2002	C	C	A	A	
	2011	C	C	A	A	
D7*						
	2002	D	D	D	D	
	2011	C	C	E	E	
D6*						
	2002	B	A	B	B	
	2011	C	C	A	A	C
D5*						
	2002	C	C	A	A	
	2011	C	C	A	A	
D3						
	2002	A	A	A	A	
	2011	B	B	A	A	

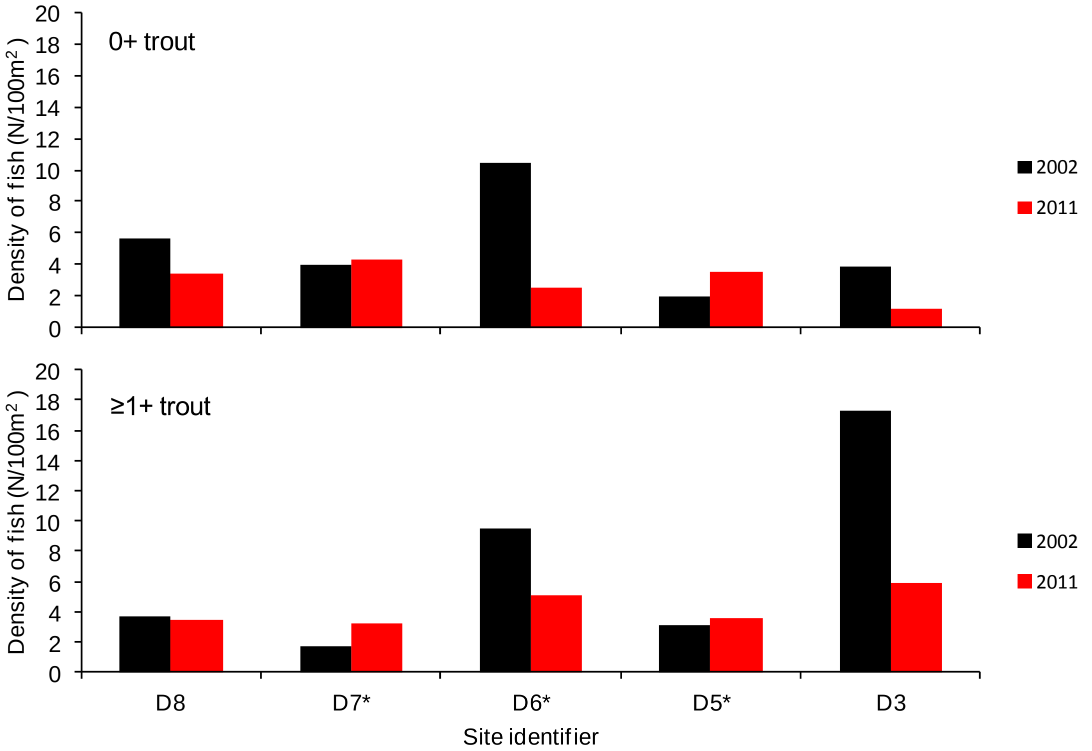

Figure 5.10 Density estimates of trout in Douglas Water in 2002 and 2011. Hydropower scheme was commissioned in 2008.

5.4.2 Salmonid size structure

Individual salmonid length data were not available from surveys carried out in 2002, therefore, it was not possible to identify if there were any changes in size structure pre and post hydropower commissioning. However, an overview of the size structure, from surveys in 2011 is provided to give an indication of size structure.

At sites D8, D7, D6 and D5 0+ trout were caught in a similar size range of 55-85 mm, but at site D3 0+ trout were in a narrow size range of 55-60 mm (Figure 5.11). ≥1+ trout were caught in a generally similar size range (105-175 mm) at sites D8, D6 and D3 but were in a narrower size range (110-145 mm) at sites D7 and D5 (Figure 5.11).

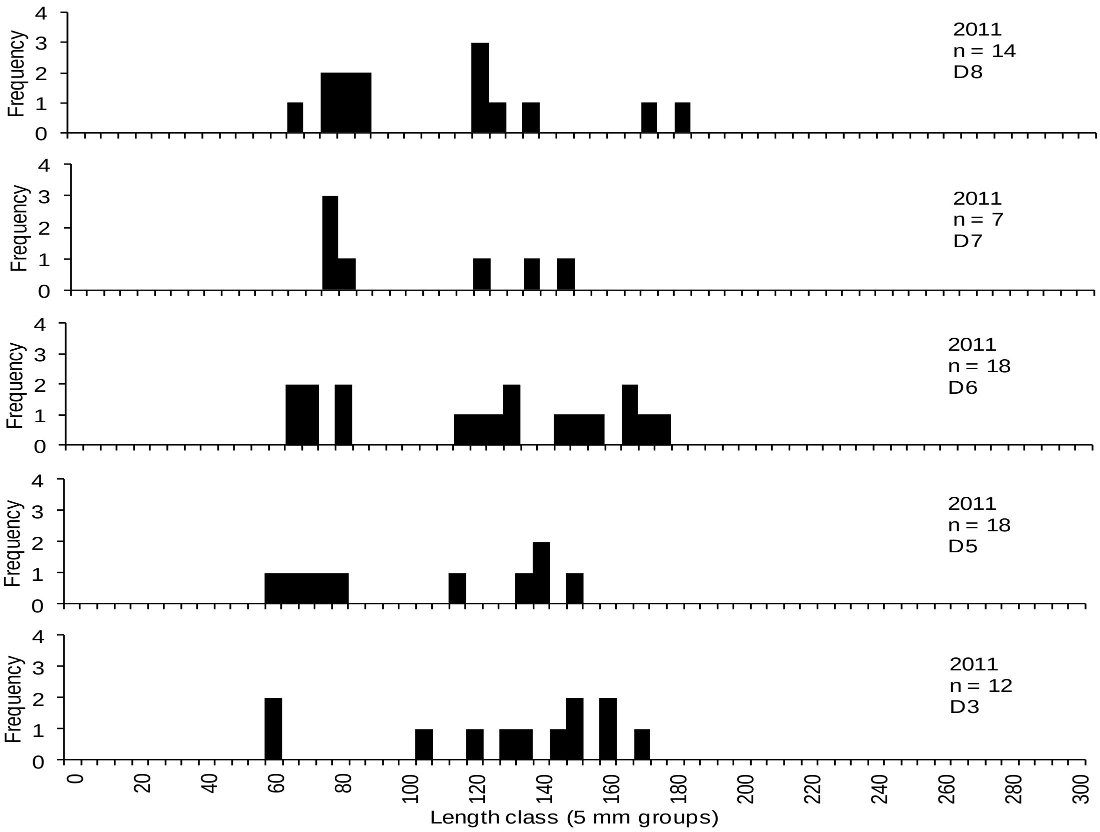

Figure 5.11 Length distributions of trout in the upper and lower reaches of Douglas Water in 2011.

5.4.3 Eel

No eels were captured in the upper or lower reaches of Douglas Water in 2002 or 2011.

5.5 Camserney Burn hydropower scheme

5.5.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

No salmon were caught in the upper reaches of Camserney Burn throughout the study period.

Upstream of the intake

0+ trout densities at site a, were low during the study period with the highest densities recorded in 2010 and 2011; 0+ trout populations were fair/poor (class D) in 2007, 2010 and 2011 and poor (class E) in other years (Figure 5.12,Table 5.10). ≥1+ trout densities varied during the study period with the highest density in 2006 and lowest in 2011; populations fluctuated between excellent (class A) and good (class B) between 2004 and 2010 but were poor (class E) in 2011 (Figure 5.12, Table 5.10).

Table 5.10 Classification of trout densities in both the upper and lower reaches of Camserney Burn between 2004 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2004	E	E	E	E	E
	2005	E	E	E	E	E
	2006	E	E	E	E	D
	2007	D	E	E	E	D
	2008	E	E	E	E	E
	2009	E	E	E	E	E
	2010	D	D	D	E	D
	2011	D	E	D	E	D
b*	2004	E	E	E	E	E
	2005	F	F	F	F	F
	2006	D	E	E	E	E
	2007	F	F	F	F	E
	2008	E	E	E	E	E
	2009	E	E	E	E	E
	2010	E	E	E	E	E
	2011	D	E	E	E	D
c*	2004	C	D	C	D	B
	2005	B	C	C	D	A
	2006	B	B	B	C	A
	2007	B	C	C	D	B
	2008	D	E	D	E	D
	2009	B	B	B	C	B
	2010	C	D	C	D	C
	2011	E	E	E	E	D
d	2004	C	C	C	D	B
	2005	B	C	B	D	A
	2006	B	C	B	C	A
	2007	B	C	C	E	B
	2008	B	C	C	D	B
	2009	C	D	C	E	C
	2010	B	C	B	D	A
	2011	C	D	D	E	C

≥1+ trout

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2004	B	B	B	B	C
	2005	B	B	B	B	C
	2006	A	A	A	A	A
	2007	A	B	A	B	B
	2008	B	C	C	C	C
	2009	B	C	C	C	C
	2010	A	B	A	B	B
	2011	E	E	E	E	E
b*	2004	D	E	E	E	E
	2005	C	C	C	C	C
	2006	A	A	A	A	A
	2007	B	B	B	B	C
	2008	B	B	B	B	C
	2009	C	C	C	D	C
	2010	C	C	C	C	C
	2011	C	C	C	D	C
c*	2004	A	A	A	A	A
	2005	A	B	B	C	B
	2006	A	A	A	A	A
	2007	A	A	A	A	A
	2008	B	B	B	B	B
	2009	B	C	B	C	C
	2010	A	A	A	A	A
	2011	A	A	A	A	A
d	2004	A	B	A	B	B
	2005	A	A	A	B	A
	2006	C	C	C	D	C
	2007	B	B	B	C	C
	2008	C	D	E	E	C
	2009	C	C	C	C	C
	2010	C	C	C	D	C
	2011	B	B	B	B	B

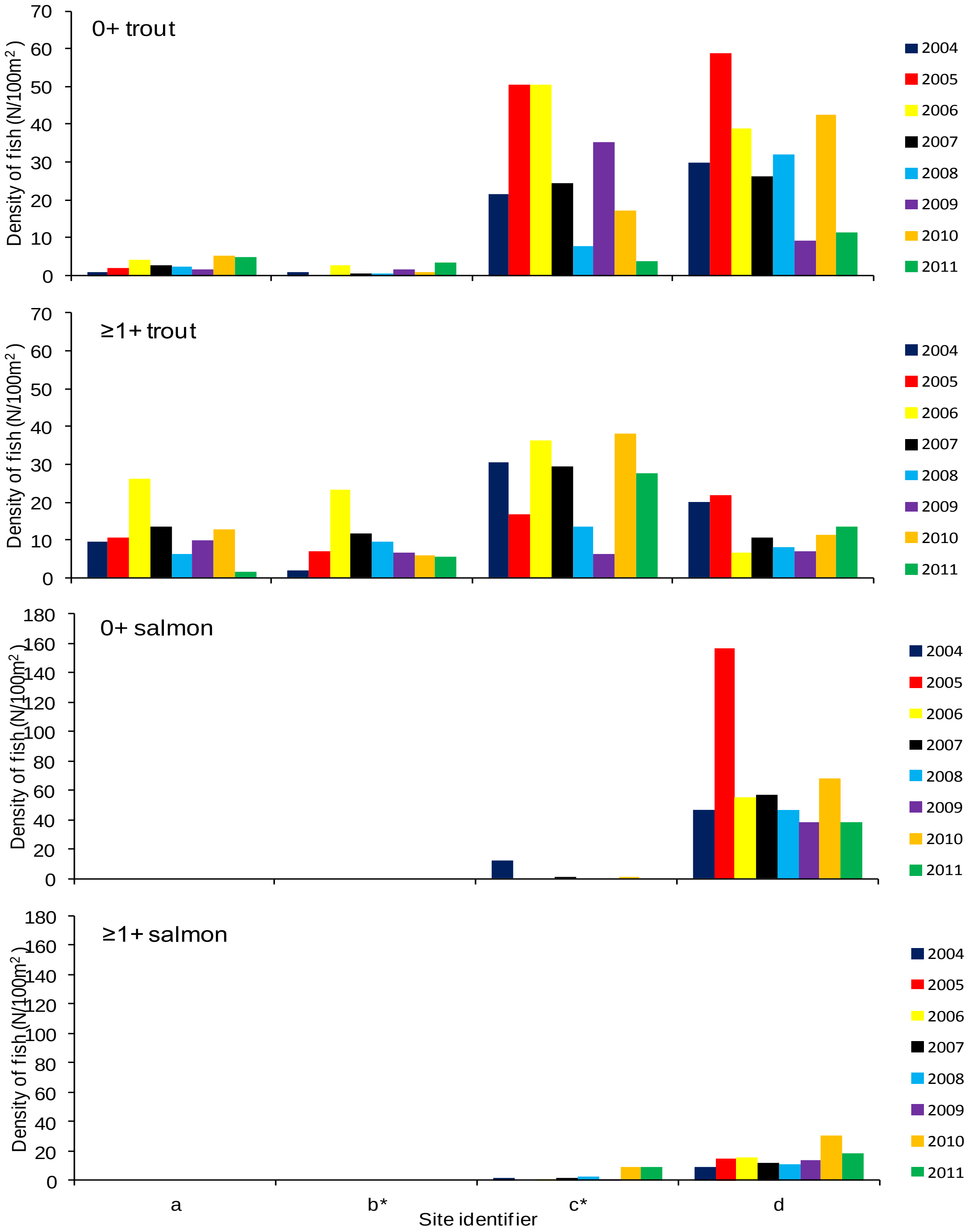

Figure 5.12 Density estimates of trout and salmon in Camserney Burn between 2004 and 2011.

Depleted reach

0+ trout populations at site b, downstream of the intake, were also low during the study period with the highest densities recorded in 2006 and 2010 similar to site a. 0+ trout populations ranged from absent (class F) in 2005, to poor (class E) and fair/average (class D) in 2006 and 2011 (Figure 5.12, Table 5.10). ≥1+ trout densities varied during the study period with the highest density in 2006 and lowest in 2004; populations fluctuated between excellent (class A) and fair/average (class C) between 2005 and

2011 but were fair/poor (class D) in 2004 (Figure 5.12, Table 5.10). ≥1+ trout populations appear to have decreased at site b since 2006, following a similar trend as found at site a upstream of the intake.

Catches of salmonids at site c, upstream of the outfall, were dominated by trout, while downstream of the outfall at site d populations were dominated by trout in some years and salmon in others (Figure 5.12).

At site c, upstream of the outfall, 0+ salmon were only captured in 2004, 2007 and 2010 and populations were poor (class E); ≥1+ salmon were absent in 2005 with the highest densities found in 2010 and 2011 when populations were fair/average (class C) (Figure 5.12, Table 5.11). The data suggest adult salmon spawning is sporadic at site c, probably reflecting the boulder dominated habitat limiting suitable salmon spawning areas. The general improving ≥1+ salmon populations at site c reflected the increasing populations found at site d downstream of the outfall (Figure 5.12).

Table 5.11 Classification of salmon densities in the lower reaches of Camserney Burn between 2005 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2004	E	E	E	E	D
	2005	F	F	F	F	F
	2006	F	F	F	F	F
	2007	E	E	E	E	E
	2008	F	F	F	F	F
	2009	F	F	F	F	F
	2010	E	E	E	E	E
	2011	F	F	F	F	F
d	2004	C	D	C	D	C
	2005	A	B	A	B	A
	2006	B	C	C	C	B
	2007	B	C	A	C	B
	2008	B	C	B	C	B
	2009	B	D	C	D	C
	2010	B	C	B	C	B
	2011	C	D	C	D	C

≥1+ salmon

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2004	E	E	E	E	E
	2005	F	F	F	F	F
	2006	E	E	E	E	E
	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2009	E	E	E	E	E
	2010	C	D	C	D	C
	2011	C	D	C	D	C
d	2004	C	C	C	C	C
	2005	B	B	B	B	B
	2006	B	B	B	B	B
	2007	C	C	B	C	B
	2008	C	C	C	C	B
	2009	C	C	C	C	B
	2010	A	B	A	B	A
	2011	B	B	B	B	B

0+ trout populations at site c fluctuated during the study period and generally mirrored patterns at site d, downstream of the outfall, except in 2008 and 2009 (Figure 5.12). 0+ trout populations at site c were highest in 2005 and 2006 and lowest in 2011 suggesting an overall decrease in populations, a trend also generally observed at site d. 0+ trout populations ranged between good (class B) in 2005, 2006, 2007 and 2009 to fair/poor (class D) in 2008 and poor (class E) in 2011 (Table 5.10). ≥1+ trout populations at site c varied during the study period with the highest densities in 2010 and lowest in 2009; populations were excellent (class A) except in 2008 and 2009 when populations were good (class B) (Figure 5.12, Table 5.10). Overall classifications of 0+ trout and ≥1+ trout populations at site c, upstream of the outfall were classified similar to or higher than populations at site d, downstream of the outfall.

Downstream of the outfall

At site d, downstream of the outfall, 0+ salmon densities were generally stable throughout the study period except in 2005 when high densities were recorded (Figure 5.12); the high density of 0+ salmon was also found in 2005 at Keltney Burn which is located a short distance further along the River Tay suggesting this year was good for salmon recruitment. 0+ salmon populations were fair/average (class C) in 2004 and 2011, excellent (class A) in 2005 and good (class B) in other years. ≥1+ salmon were similar between 2004 and 2009 with the highest density in 2010 and an overall increasing trend (Figure 5.12); populations ranged from fair/average (class C) to excellent (class A) (Table 5.11).

0+ trout populations at site d, showed an overall decrease, except in 2010, during the study period, as found at site c, and were classified as good (class B) except in 2004, 2009 and 2011 when populations were fair/average (class C) (Figure 5.12, Table 5.10). ≥1+ trout populations, were highest in 2005 but were generally stable in subsequent years; populations were excellent (class A) in 2004 and 2005 and varied between good (class B) and fair/average (class C) in other years (Figure 5.12, Table 5.10).

5.5.2 Salmonid size structure

Length distributions of brown trout in the upper reaches associated with the intakes varied between years at each site.

Upstream of the intake

At site a, upstream of the intake, 0+ trout were caught in generally low numbers and overall in a similar size range between 45-75 mm (Figure 5.13). ≥1+ trout were caught in the size range 75-180 mm in 2006, 2007, 2009 and 2010 and a wider size range of 80-200 mm in 2004 and 2005 (Figure 5.13). ≥1+ trout were caught in a narrow size range of 85-145 mm in 2008 and in low numbers in 2011 and in a very narrow size range (Figure 5.13).

Depleted reach

At site b, downstream of the intake, 0+ trout were caught in low numbers throughout the study period and in a generally narrow size range of 60-75 mm (Figure 5.14). ≥1+ trout were caught in a wide size range (80-200 mm) in 2006 and narrower size range in other years (Figure 5.14).

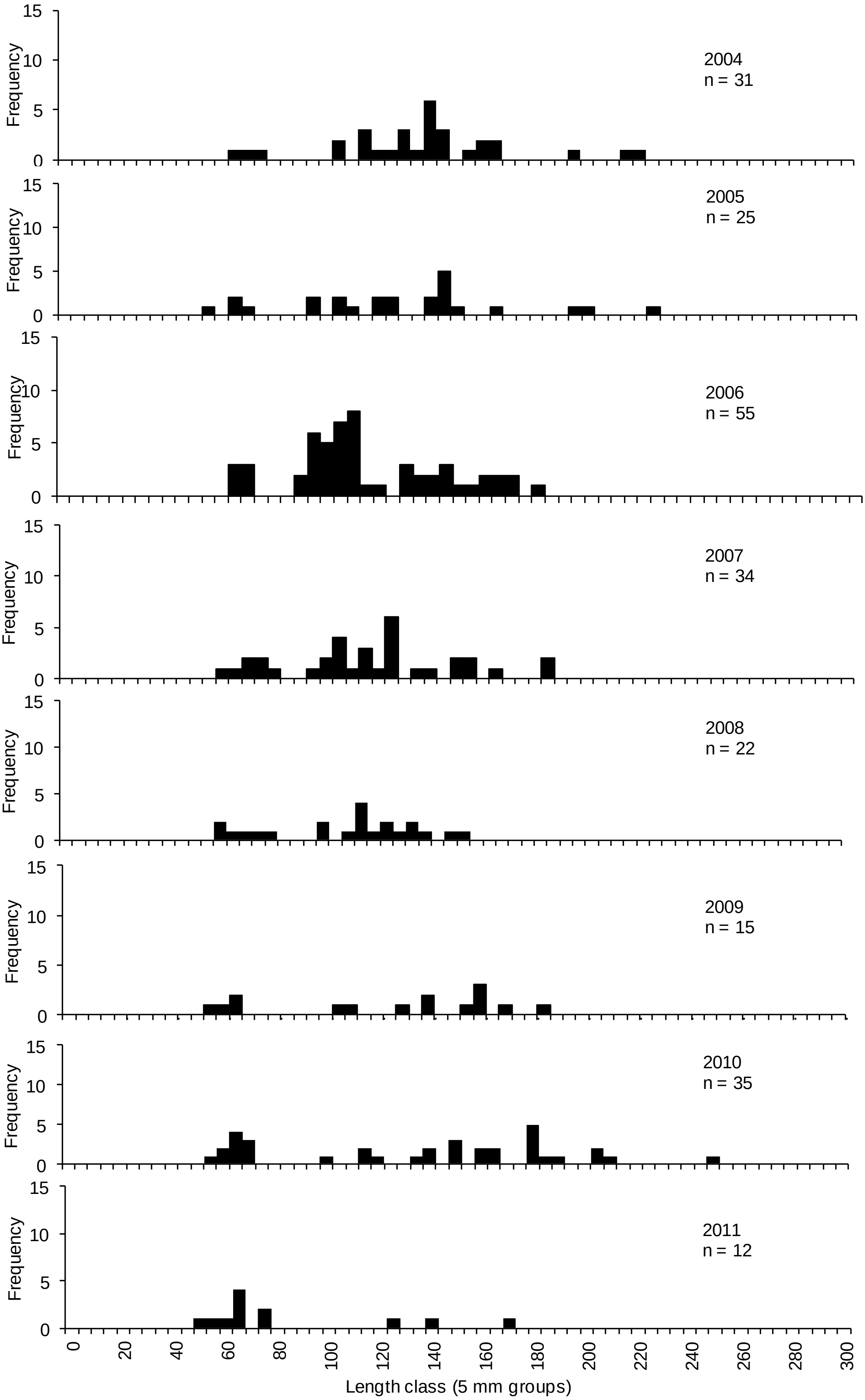

Figure 5.13 Length distributions of trout at site a in Camserney Burn between 2004 and 2011.

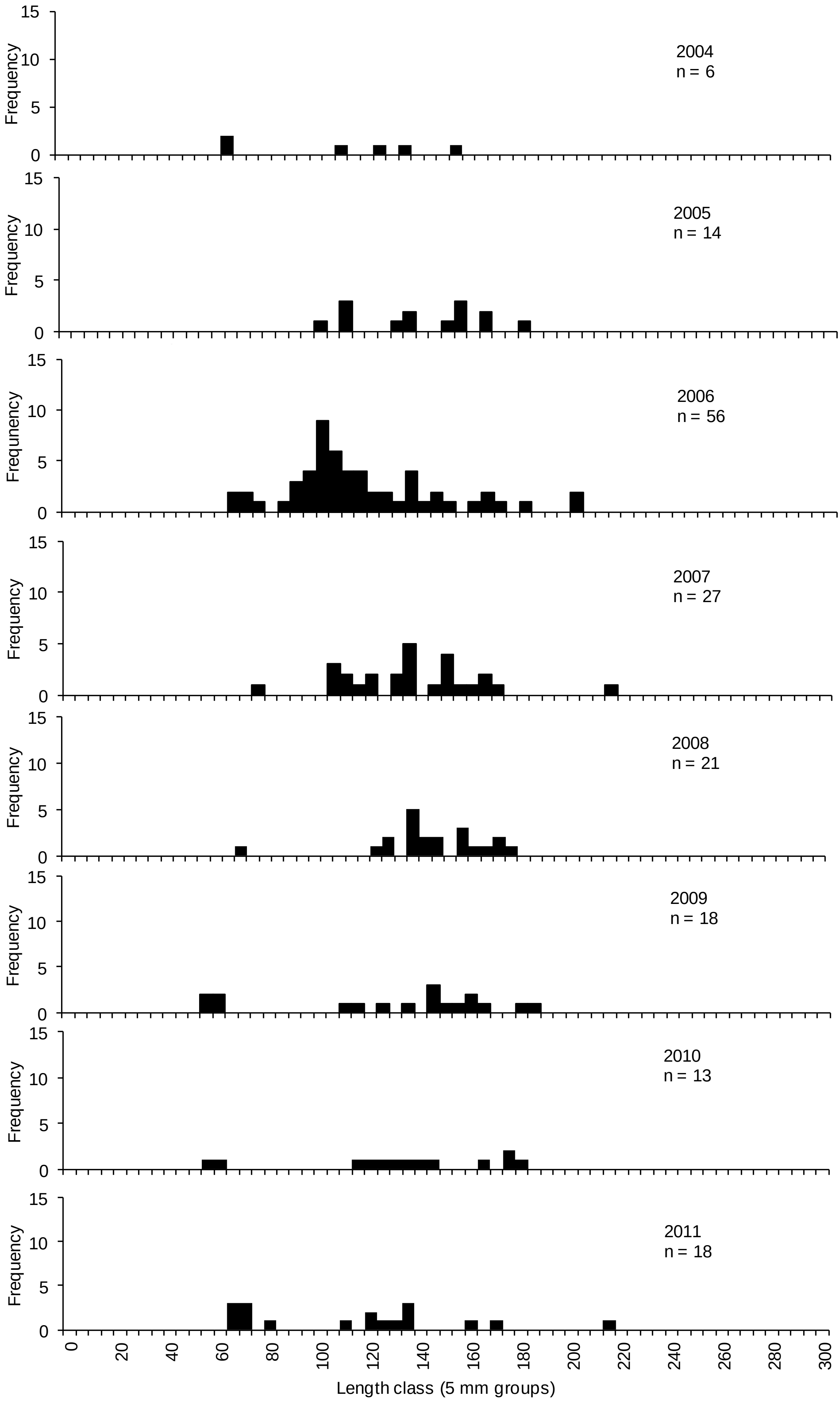

Figure 5.14 Length distributions of trout at site b in Camserney Burn between 2004 and 2011.

At site c, upstream of the outfall, 0+ salmon were caught in low numbers throughout the study period and in a generally narrow size range except in 2004 (Figure 5.15). ≥1+ salmon were caught in a wide size range (80-130 mm) in 2011 and narrower size range in other years (Figure 5.15). At site c 0+ trout were caught in a similar size range of 35-75 mm in most years except in 2008 and 2011 when 0+ trout were in a narrower size range (Figure 5.17). ≥1+ trout were caught in a wide size range (80-200 mm) in most years with one trout >200 mm caught in 2005 (Figure 5.17).

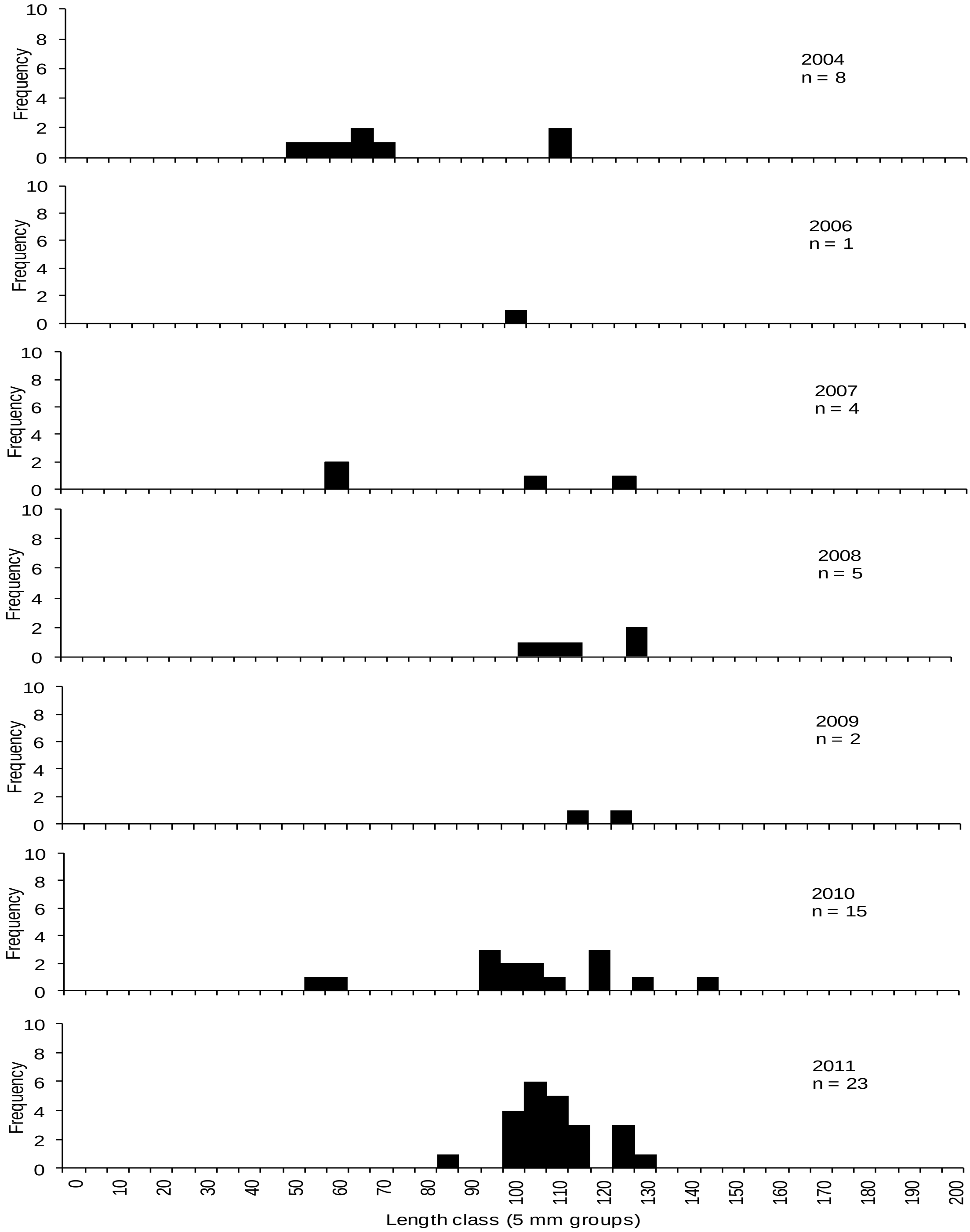

Figure 5.15 Length distributions of salmon at site c in Camserney Burn between 2004 and 2011. Note no salmon were captured at site c in 2005.

Downstream of the outfall

Length distributions of salmon at site d, downstream of the outfall, were broadly similar between years, albeit in lower numbers in some years (Figure 5.16). 0+ salmon were mainly in the size range 35-70 mm and ≥1+ salmon were mainly in the size range 80-120 mm (Figure 5.16). 0+ trout were caught in the size range 35-75 mm in most years, albeit in varying numbers; ≥1+ trout were caught in a similar size range of 85-180 mm with larger individuals >180 mm captured in 2005, 2009 and 2010 (Figure 5.18).

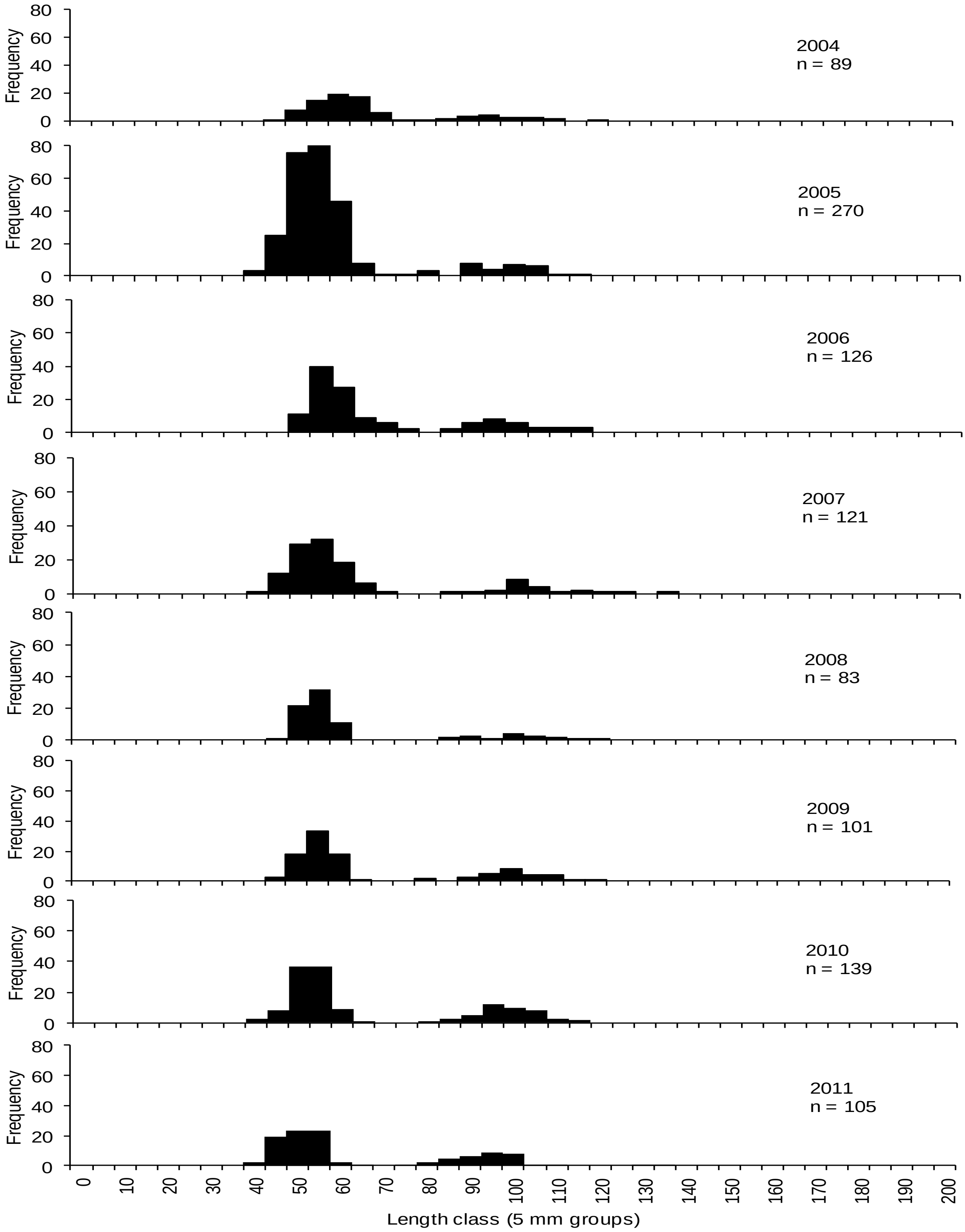

Figure 5.16 Length distributions of salmon at site d in Camserney Burn between 2004 and 2011.

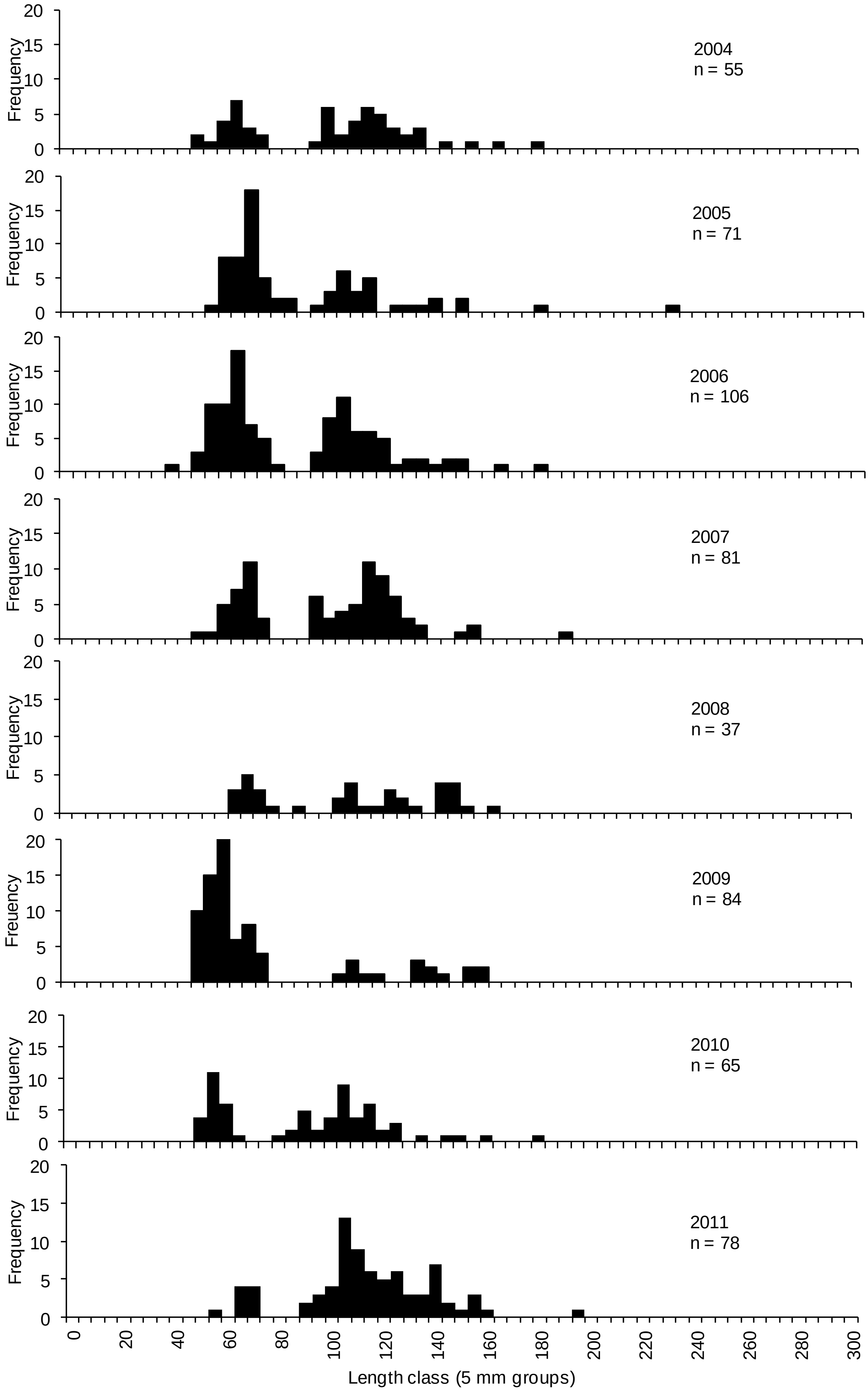

Figure 5.17 Length distributions of trout at site c in Camserney Burn between 2004 and 2011.

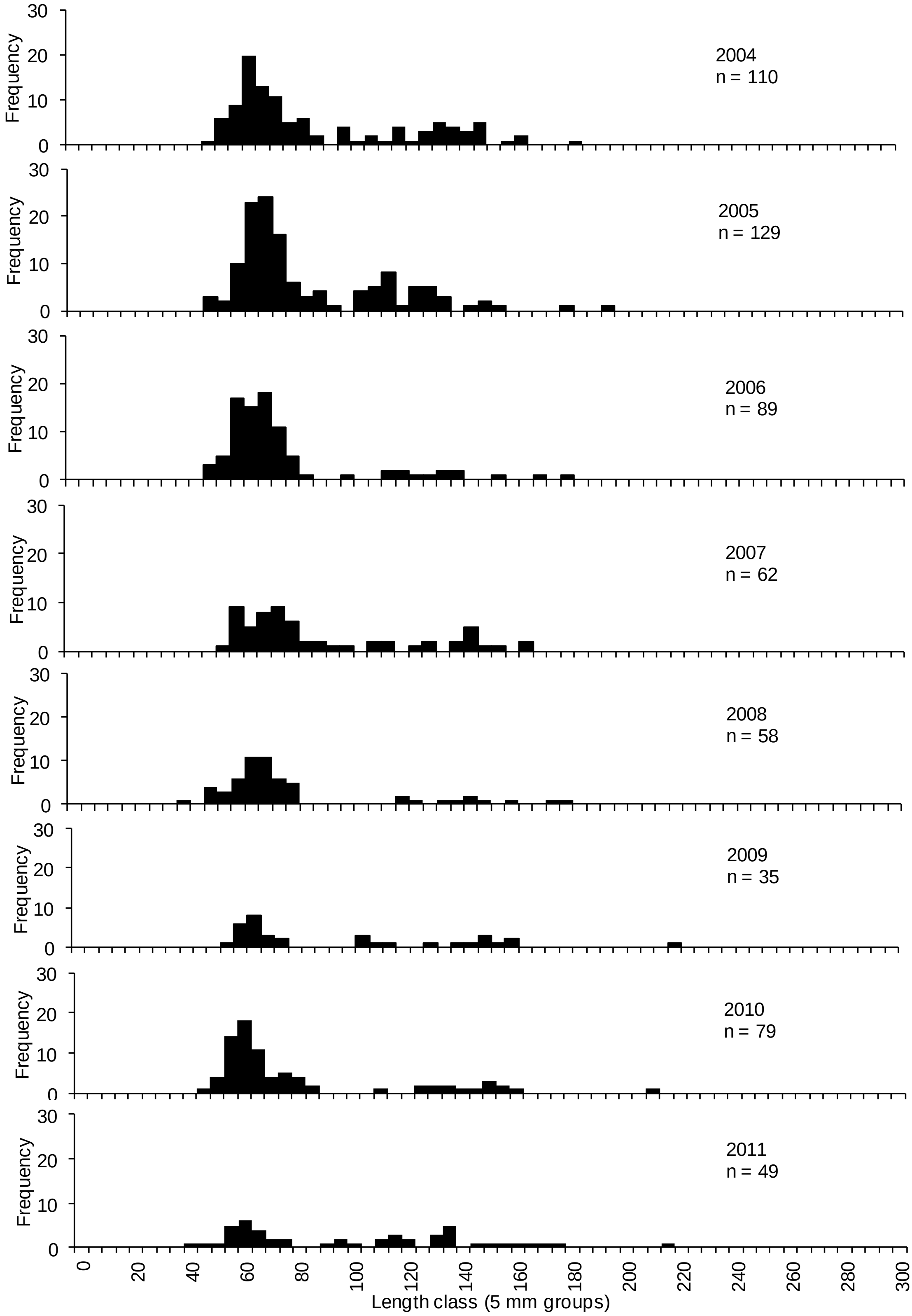

Figure 5.18 Length distributions of trout at site d in Camserney Burn between 2004 and 2011.

5.5.3 Salmonid growth rates

Upstream of the intake

At site a, back-calculated length of trout at age 1 ranged between 58 and 84 mm with slowest growth in 2002 and 2003 and fastest growth in 2009 (Table 5.12). Back-calculated length of trout at age 2 ranged between 105 and 137 mm, with the slowest growth in 2002 and fastest growth in 2008 (Table 5.12). Back-calculated length of trout at age 3 ranged between 147 and 164 mm; back-calculated length of trout at age 4 was 211 mm.

Table 5.12. Back-calculated lengths at age of trout in the upper and lower reaches of Camserney Burn at sites a-d. Note ageing data from 2004 was unavailable. * see Section 3.3.4 for details

Site code	Year class	Back-calculated length at age (mm ± S.D (n))				
		1	2	3	4	5
a	2001	67±7(2)	109±4(2)	164±6(2)	211±15(2)	
	2002	58±6(5)	105±10(5)	147±9(5)		
	2003	58±11(9)	109±12(9)			
	2004	63±7(4)				
	2006	60±7(4)	112±2(4)			
	2007	61±7(15)	136±6(3)			
	2008	69±8(9)	137±16(6)			
	2009	84±15(12)	129(1)			
	2010	73±6(2)				
	*2011	62±7(9)				
b	2001	55±3(3)	112±3(3)	152±7(3)	185±10(3)	
	2002	60±10(11)	113±12(11)	151±13(11)		
	2003	61±10(8)	110±13(8)	140±9(2)		
	2004	53±4(11)	101±10(11)			
	2005	54±8(11)	110±1(2)	146±1(2)		
	2006	56±9(8)	110±11(8)	162(1)		
	2007	55±6(6)	115±13(5)			
	2008	70±11(6)	134±4(3)			
	2009	80±13(9)	125±3(2)			
	2010	68±5(7)				
	*2011	67±5(7)				
c	2000	61(1)	95(1)	156(1)	189(1)	230(1)
	2001	53(1)	94(1)	142(1)	177(1)	
	2002	58±11(4)	109±6(4)	143±6(4)		
	2003	58±8(11)	107±11(11)	135(1)		
	2004	60±8(16)	103±6(8)			
	2005	57±6(9)	89(1)	134(1)		
	2006	55±4(3)	108±5(3)			
	2007	63±12(12)	88(1)	132(1)		
	2008	63±4(3)	121±13(3)			
	2009	69±10(27)	124±18(5)			
	2010	64±9(16)				
	*2011	63±4(9)				
d	2000	63(1)	120(1)	155(1)	172(1)	195(1)
	2001	45(1)	111(1)	150(1)	178(1)	
	2002	51±10(3)	99±10(3)	148±6(3)		
	2003	54±9(13)	116±16(13)	140(1)		
	2004	69±9(11)	116±15(4)			
	2005	58±12(10)	112(1)	144(1)		
	2006	60±9(5)	113±13(5)			
	2007	55±10(12)	116±11(7)			
	2008	64±14(6)	149±22(2)	182(1)		
	2009	82±14(19)	126±11(5)			
	2010	70±8(13)				
	*2011	58±9(22)				

Depleted reach

At site b, back-calculated length of trout at age 1 ranged between 53 and 80 mm with slowest growth in 2004 and fastest growth in 2009 (Table 5.12). Back-calculated length of trout at age 2 ranged between 101 and 134 mm, with the slowest growth in 2004 and fastest growth in 2008 (Table 5.12). Back-calculated length of trout at age 3 ranged between 140 and 162 mm, with the slowest growth in 2003 and fastest in 2001; back-calculated length of trout at age 4 was 185 mm (Table 5.12).

At site c, back-calculated length of trout at age 1 ranged between 53 and 69 mm with slowest growth in 2001 and fastest growth in 2009 (Table 5.12). Back-calculated length of trout at age 2 ranged between 88 and 124 mm, with the slowest growth in 2007 and fastest growth in 2009 (Table 5.12). Back-calculated length of trout aged 3 ranged between 132 and 156 mm, with the slowest growth in 2007 and fastest growth in 2000 (Table 5.12). Back-calculated length of trout at age 4 ranged from between 177 and 189 mm, and at age 5 was 230 mm. At site c back-calculated length of salmon at age 1 ranged between 52 and 65 mm with slowest growth in 2006 and fastest in 2009 (Table 5.13). Back-calculated length of salmon at age 2 ranged between 95 and 106 mm, with the slowest growth in 2009 and fastest growth in 2008 (Table 5.13).

Table 5.13 Back-calculated lengths at age of salmon in the lower reaches of Camserney Burn at sites c-d. Note no 0+ salmon were captured in 2011 at site c and ageing data from 2004 was unavailable. *see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))				
		1	2	3	4	5
c	2006	52±2(2)	101±9(2)			
	2007	59±7(2)				
	2008	64±5(2)	106±3(2)			
	2009	65±11(14)	95±1(3)			
	2010	63±7(9)				
d	2002	46(1)	83(1)	115(1)		
	2003	46±8(6)	97±9(6)			
	2004	58±4(6)				
	2005	50±7(12)				
	2006	49±3(2)	98±2(2)			
	2007	61±5(9)				
	2008	56±8(11)				
	2009	64±5(16)				
	2010	61±11(11)				
	*2011	49±5(70)				

Downstream of the outfall

At site d, back-calculated length of trout at age 1 ranged between 45 and 82 mm with slowest growth in 2001 and fastest in 2009 (Table 5.12). Back-calculated length of trout at age 2 ranged between 99 and 149 mm, with the slowest growth in 2002 and fastest growth in 2008 (Table 5.12). Back-calculated length of trout aged 3 ranged between 140 and 182 mm, with the slowest growth in 2003 and fastest growth in 2008 (Table 5.12). Back-calculated length of trout at age 4 ranged between 172 and 178 mm, and at age 5 was 195 mm.

At site d, back-calculated length of salmon at age 1 ranged between 46 and 64 mm with slowest growth in 2002 and 2003 and fastest growth in 2009 (Table 5.13). Back-calculated length of salmon at age 2 ranged between 83 and 98 mm, with the slowest growth in 2002 and fastest in 2006 (Table 5.13). Back-calculated length of salmon at age 3 was 115 mm.

5.5.4 Eel

No eels were captured in the upper reaches of Camserney Burn throughout the study period. Numbers of eels captured at sites c and d varied between survey years and data suggest the lower reaches of Camserney Burn are used by small numbers of eels (Figure 5.19, Table 5.14).

Table 5.14 Number of eel captured at survey sites in the lower reaches of Camserney Burn between 2004 and 2011.

Year	Site identifier	
	c	d
2004	3	5
2005	3	12
2006	0	11
2007	5	8
2008	5	2
2009	5	3
2010	13	8
2011	13	6

At site c, upstream of the outfall, eel numbers improved while at site d, downstream of the outfall eel numbers fluctuated between twelve in 2005 and two in 2008 (Table 5.14). Despite variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers with the hydropower scheme operational.

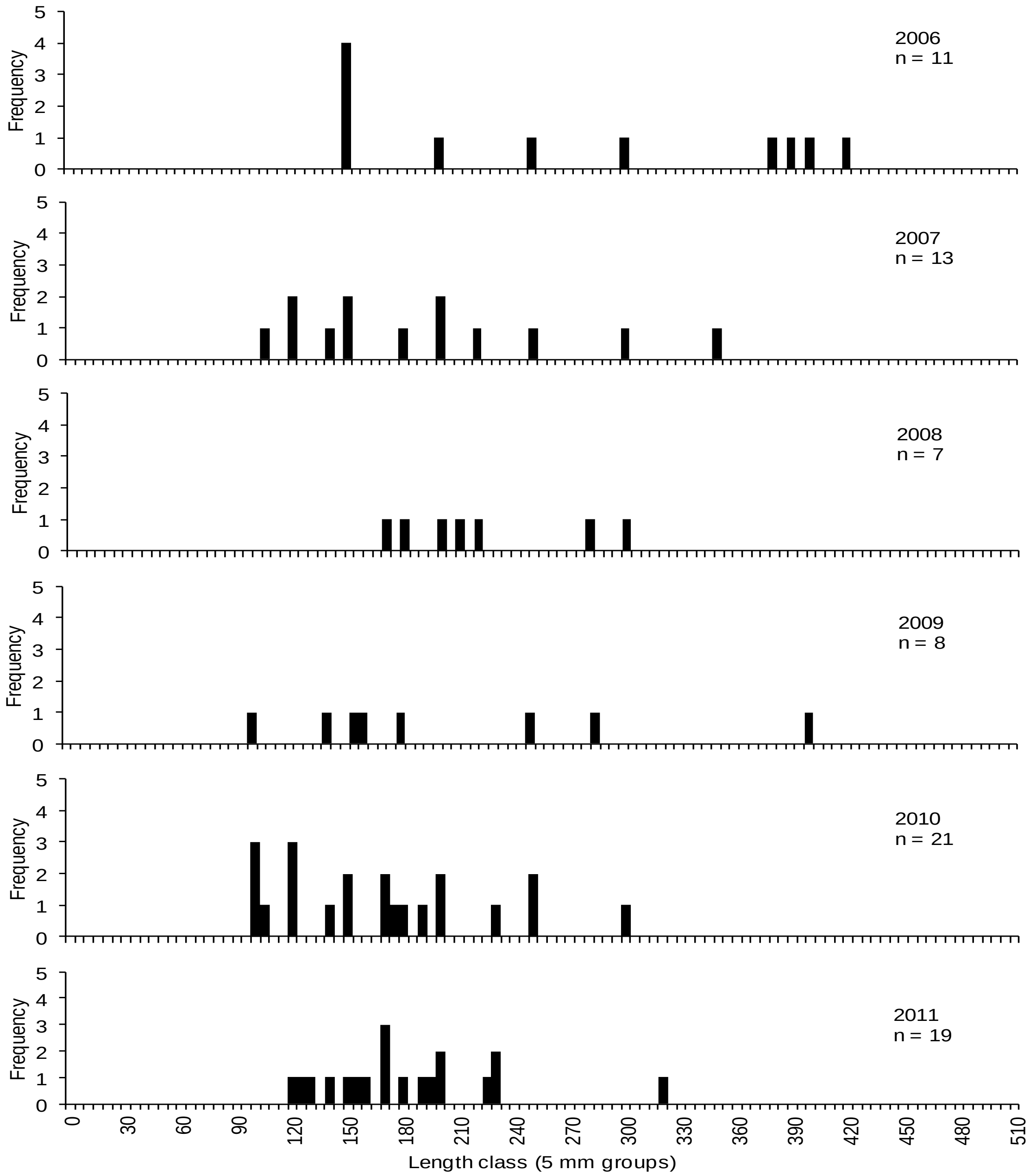

Figure 5.19 Combined length distributions of eels from sites c-d in Camserney Burn between 2004 and 2011. Note eel raw lengths in 2005 were not recorded.

5.6 Allt Gleann Da-Eig hydropower scheme

5.6.1 Salmonid population trends

The following descriptions of the density classifications are based on the national SFCC scheme; variations were often found when analysing data using the other classification schemes.

No salmon were captured in the upper reaches of Allt Gleann Da-Eig associated with the intake except in 2007 when two ≥1+ salmon were caught. It is likely that salmon were stocked due to waterfalls downstream of the survey sites being impassable to adult salmon and thus precluding natural colonisation (HIFI, pers. comm.).

0+ trout densities at site a, upstream of the intake, were low during the study period and classified as poor (class E) (Figure 5.20, Table 5.15). ≥1+ trout densities at site a were also low throughout the study period ranging from fair/average (class C) in 2010 to poor (class E) in 2007 and 2008.

Table 5.15 Classification of trout densities in both the upper and lower reaches of Allt Gleann Da-Eig between 2007 and 2011 based on five classification schemes. Note hydropower scheme commissioned October 2011. Site identifiers marked with * indicate location within abstracted reach.

0+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2010	E	E	E	E	E
	2011	E	E	E	E	E
b*	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2010	E	E	E	E	E
	2011	F	F	F	F	F
c*	2007	n/s	n/s	n/s	n/s	n/s
	2008	D	E	D	E	D
	2010	D	E	C	D	D
	2011	F	F	F	F	F
d*	2007	E	E	E	E	E
	2008	D	E	D	E	C
	2010	D	D	D	E	D
	2011	E	E	E	E	E
e	2007	F	F	F	F	F
	2008	E	E	E	E	E
	2010	E	E	E	E	D
	2011	E	E	E	E	D

≥1+ trout

Site/Year		SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
a	2007	E	E	D	E	E
	2008	E	E	E	E	E
	2010	C	C	B	B	C
	2011	D	E	D	E	D
b*	2007	D	D	E	E	D
	2008	D	D	D	D	D
	2010	B	C	B	C	C
	2011	E	E	C	C	E
c*	2007	n/s	n/s	n/s	n/s	n/s
	2008	D	D	D	E	D
	2010	D	D	D	D	D
	2011	D	E	C	B	D
d*	2007	D	D	D	E	D
	2008	E	E	D	E	D
	2010	C	C	C	C	C
	2011	C	C	B	B	C
e	2007	E	E	E	E	E
	2008	F	F	F	F	E
	2010	D	D	C	D	D
	2011	E	E	E	E	E

0+ trout densities at site b, downstream of the intake, were low or zero during the study period and classified as poor (class E) or absent (class F) (Figure 5.20, Table 5.15). ≥1+ trout densities at site b were also low except in 2010 ranging from good (class B) in 2010 to poor (class E) in 2011.

At site c, upstream of the outfall and road bridge, 0+ salmon were absent in 2008, 2010 and 2011, while ≥1+ salmon populations were low or zero and classed as poor (class E) or absent (class F) (Figure 5.20, Table 5.16). ≥1+ trout populations were stable between 2008 and 2011 and fair/poor (class D) (Figure 5.20, Table 5.15).

At site d, upstream of the outfall and downstream of the road bridge, 0+ salmon were only captured in 2007 and populations were poor (class E), while ≥1+ salmon populations were low or zero and classed as poor (class E) or absent (class F) (Figure 5.20, Table 5.16). 0+ trout at site d were only captured in low densities in 2007 and 2011 and higher densities in 2008 and 2010; 0+ trout populations ranged from fair/poor (class D) to poor (class E) (Figure 5.20, Table 5.15). ≥1+ trout populations were lowest in 2008 and highest in 2010 and ranged between fair/average (class C) in 2010 and 2011 to poor (class E) in 2008.

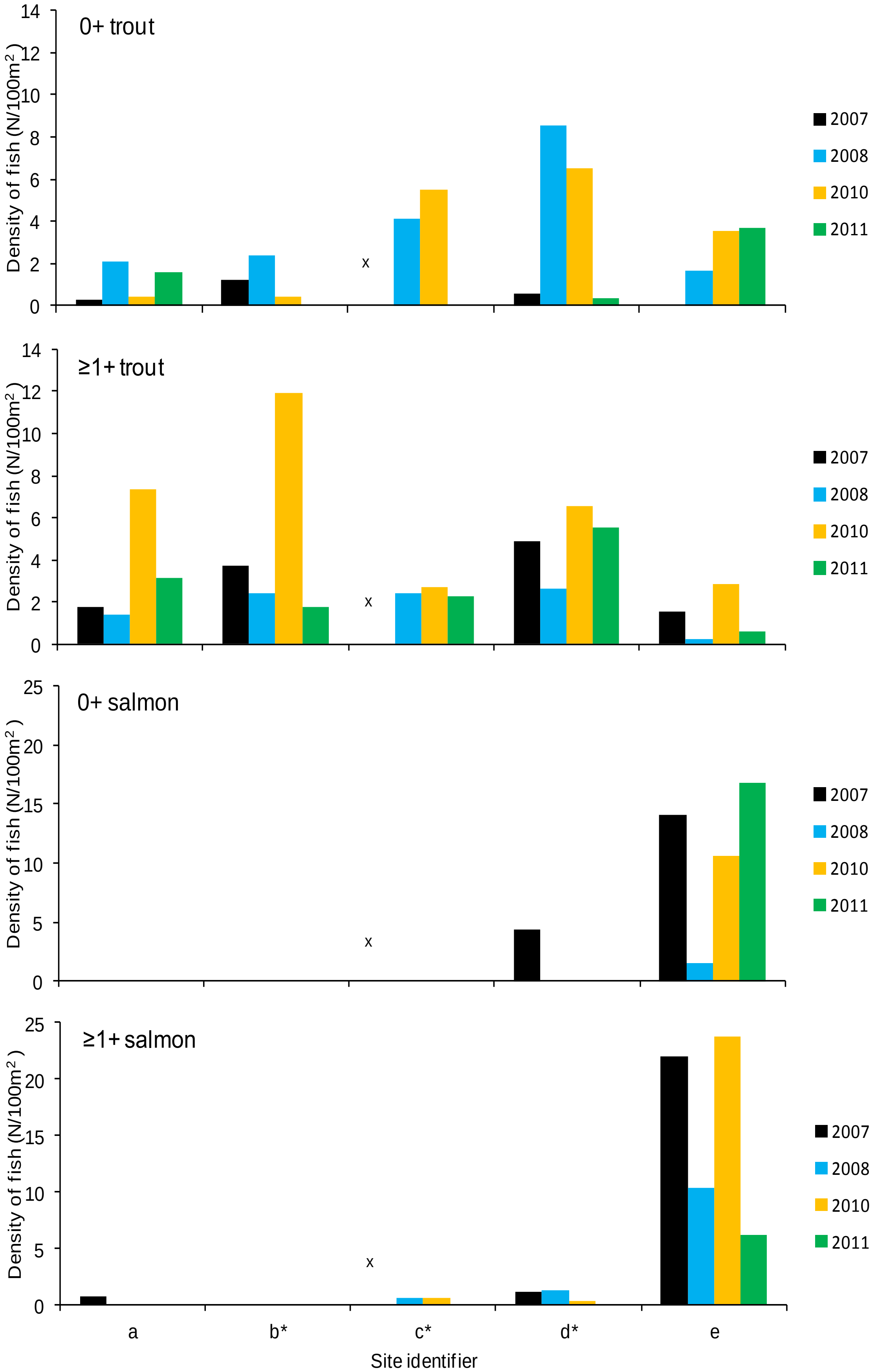

Figure 5.20 Density estimates of trout and salmon in Allt Gleann Da-Eig between 2007-2008 and 2010-2011. Note site c was not sampled in 2007 and all sites were not sampled in 2009. Hydropower scheme was commissioned October 2011.

Table 5.16 Classification of salmon densities in the lower reaches of Allt Gleann Da-Eig between 2007 and 2011 based on five classification schemes. Site identifiers marked with * indicate location within abstracted reach.

0+ salmon

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2007	n/s	n/s	n/s	n/s	n/s
	2008	F	F	F	F	F
	2010	F	F	F	F	F
	2011	F	F	F	F	F
d*	2007	E	E	E	E	E
	2008	F	F	F	F	F
	2010	F	F	F	F	F
	2011	F	F	F	F	F
e	2007	D	D	D	E	D
	2008	E	E	E	E	E
	2010	D	E	D	E	D
	2011	D	E	D	E	D

≥1+ salmon

Site	Year	SFCC National	SFCC Regional	National R.Width	Regional R.Width	EA-FCS
c*	2007	n/s	n/s	n/s	n/s	n/s
	2008	E	E	E	E	E
	2010	E	E	E	E	E
	2011	F	F	F	F	F
d*	2007	E	E	E	E	E
	2008	E	E	E	E	E
	2010	E	E	E	E	E
	2011	F	F	F	F	F
e	2007	B	B	B	C	A
	2008	D	D	D	D	B
	2010	B	B	B	B	A
	2011	E	E	E	E	C

At site e, downstream of the outfall, 0+ salmon densities were lowest in 2008 and highest in 2011 and populations were predominantly fair/average (class D); ≥1+ salmon densities were highest in 2010 and lowest in 2011 ranging from good (class B) to poor (class E) (Figure 5.20, Table 5.16). 0+ trout were absent in 2007 with densities generally improving during the study period but remaining poor (class E); ≥1+ trout populations were lowest in 2008 and highest in 2010 and ranged between absent (class F) and fair/average (class D) (Figure 5.20, Table 5.15).

5.6.2 Salmonid size structure

Two salmon in the 135-140 mm size range were caught at site a in 2007.

Length distributions of brown trout in the upper reaches associated with the intakes varied between years at each site. At site a, upstream of the intake, 0+ trout were caught in generally low numbers and overall in a similar size range between 45-75 mm (Figure 5.21). ≥1+ trout were caught in the size range 80-160 mm in 2010 and a narrower size range in other years (Figure 5.21).

At site b, downstream of the intake, 0+ trout were caught in low numbers throughout the study period and in a generally narrow size range of 50-75 mm in 2008 and a narrower size range in other years (Figure 5.22). ≥1+ trout were caught in a wide size range (80-180 mm) in 2010 and narrower size range in other years (Figure 5.22).

At site c, upstream of the outfall, 0+ trout were caught in the size range 50-75 mm in 2008 and 2010, while ≥1+ trout were caught in a wide size range but in low numbers (Figure 5.23).

At site d, upstream of the outfall, 0+ salmon were caught in the size range 50-75 mm in 2007 while ≥1+ salmon were caught in a narrow size range and in low numbers (Figure

5.24). At site d 0+ trout were caught in a similar size range of 35-80 mm in 2008 and 2010 and in low numbers in other years; ≥1+ trout were caught in a wide size range (100-200 mm) in most years with one trout >200 mm caught in 2007 and 2008 (Figure 5.25).

At site e, downstream of the outfall, 0+ salmon were caught in a similar size range 35-65 mm in all years while ≥1+ salmon were caught in the size range 75-120 mm (Figure 5.26). At site e 0+ trout were caught in a similar size range of 45-80 mm in 2008, 2010 and 2011 and were absent in 2007; ≥1+ trout were caught in a narrow size range in all years (Figure 5.27).

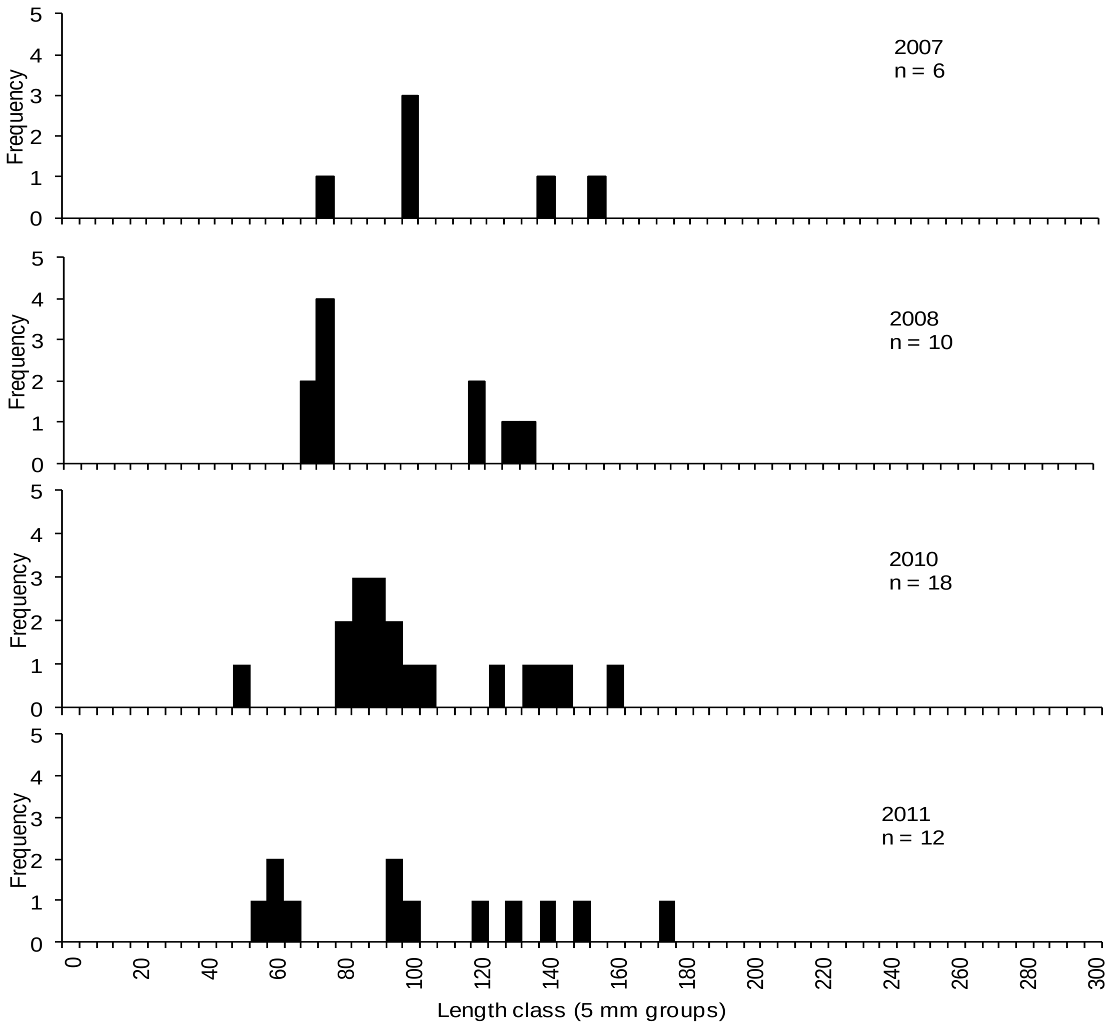

Figure 5.21 Length distributions of trout at site a in Allt Gleann Da-Eig between 2007 and 2011.

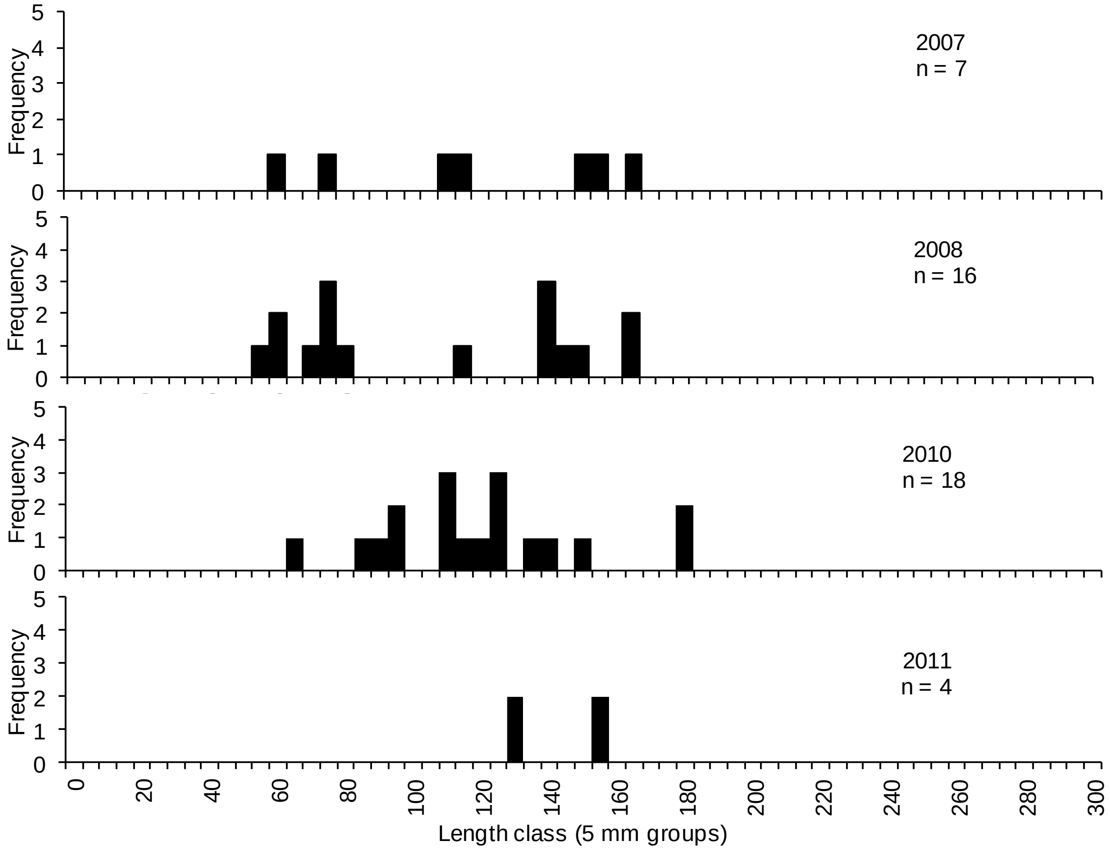

Figure 5.22 Length distributions of trout at site b in Allt Gleann Da-Eig between 2007 and 2011.

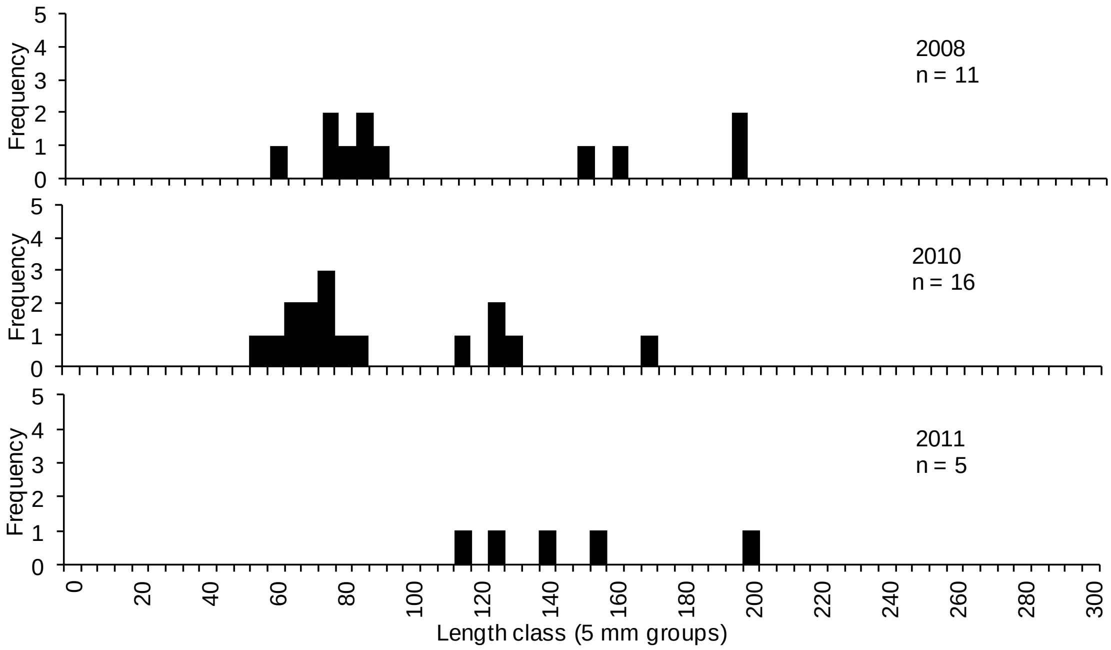

Figure 5.23 Length distributions of trout at site c in Allt Gleann Da-Eig between 2008 and 2011. Note site c was not surveyed in 2007.

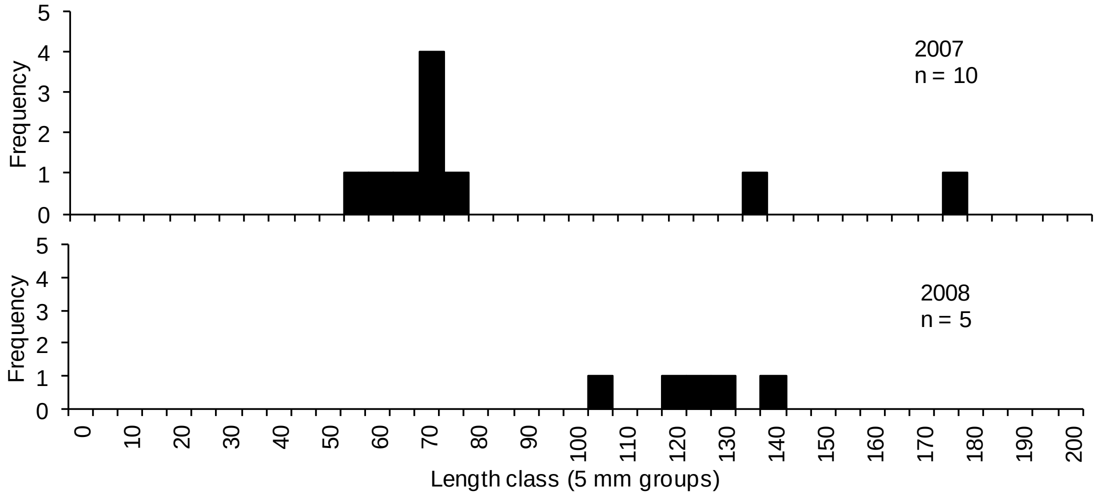

Figure 5.24 Length distributions of salmon at site d in Allt Gleann Da-Eig between 2007 and 2008. Note one salmon was caught in 2010.

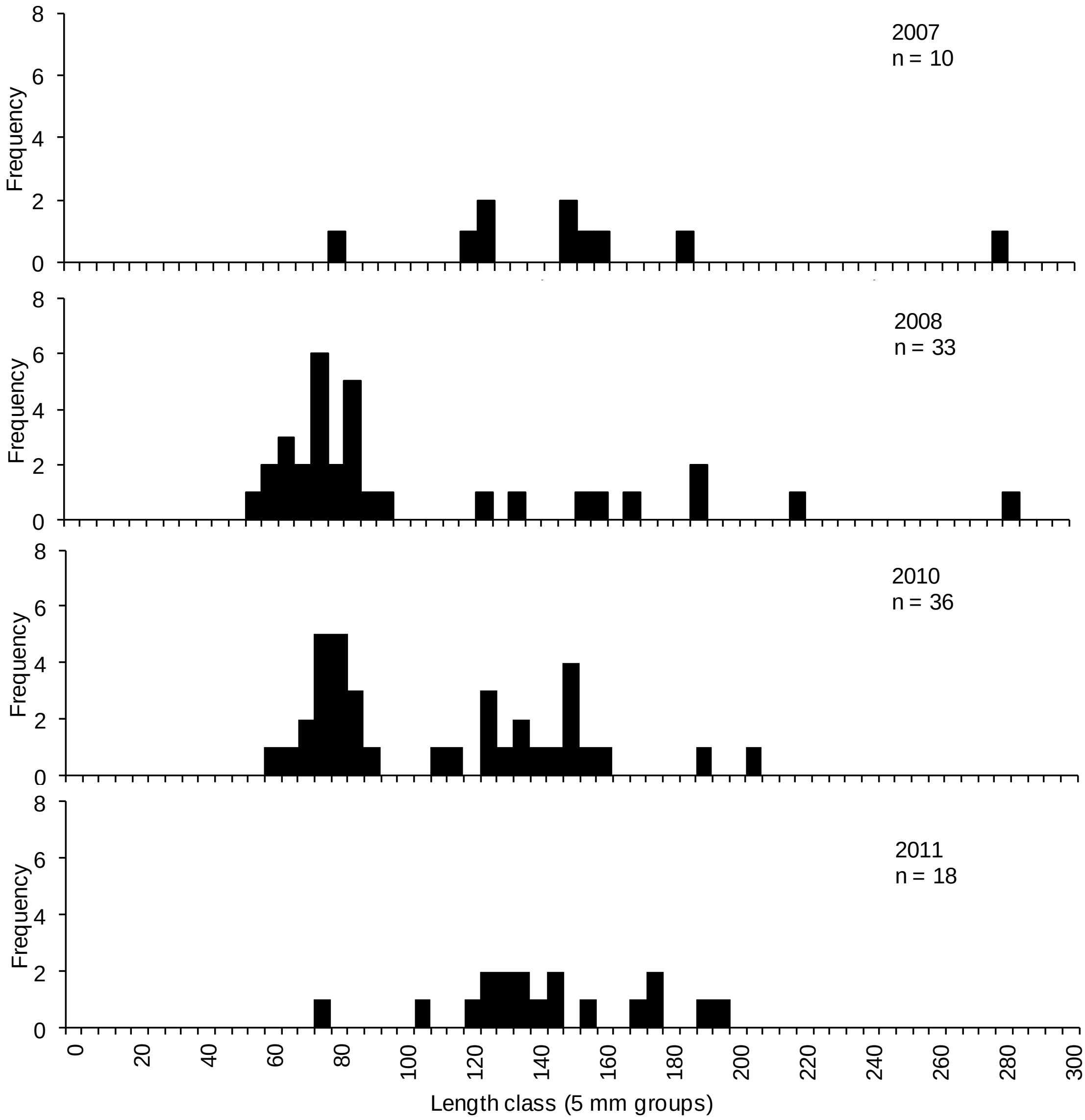

Figure 5.25 Length distributions of trout at site d in Allt Gleann Da-Eig between 2007 and 2011.

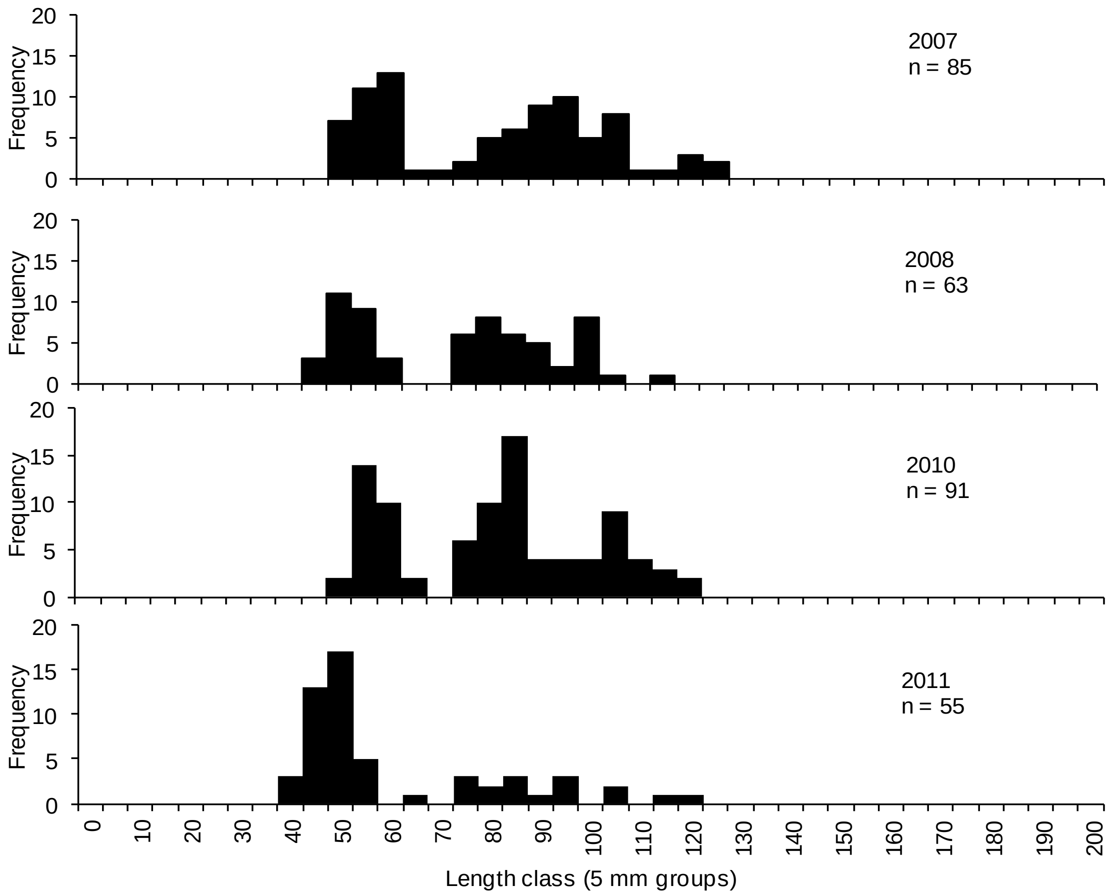

Figure 5.26 Length distributions of salmon at site e in Allt Gleann Da-Eig between 2007 and 2011.

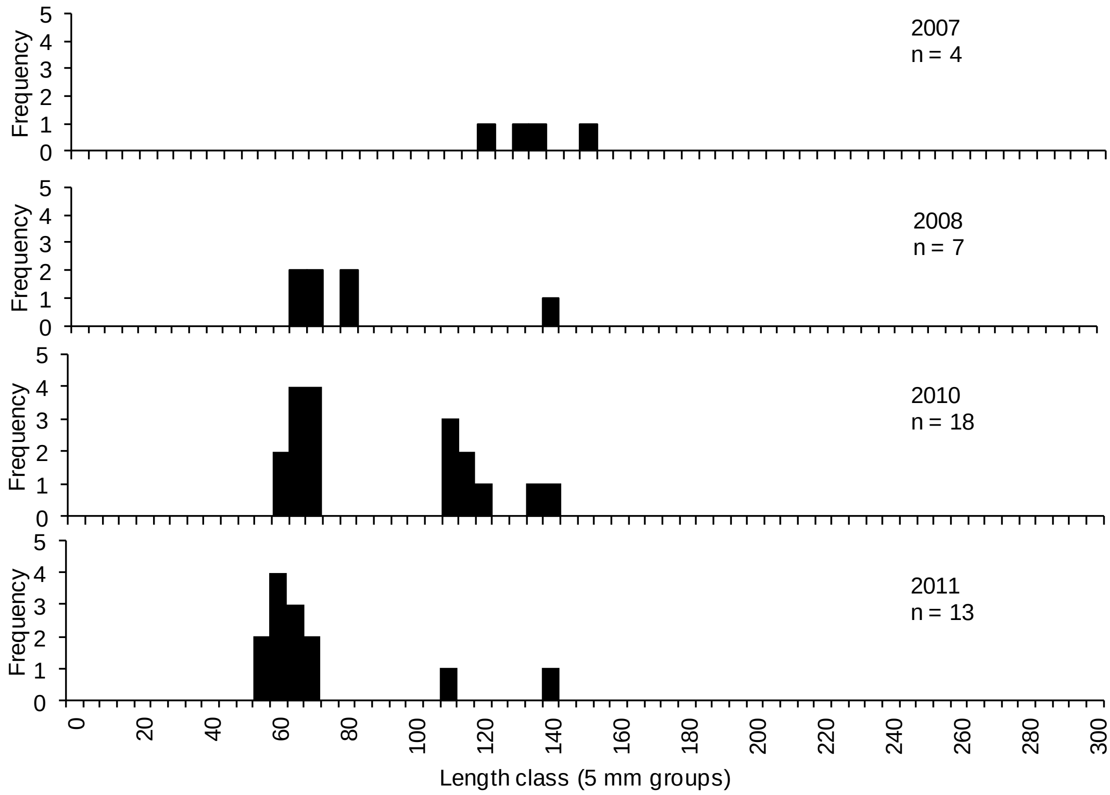

Figure 5.27 Length distributions of trout at site e in Allt Gleann Da-Eig between 2007 and 2011.

5.6.3 HABSCORE analysis

Habitat parameters and fish population data collected before the hydropower scheme was constructed were compared to the same information collected following construction at two key sites in the lower reaches of Allt Gleann Da-Eig. For the analysis habitat data collected in 2008 were utilised with fisheries data collected from 2007-2008 was compared to habitat data collected in 2011 utilised with fisheries data collected from 2010 and 2011 (Tables 5.17 & 5.18).

The observed density of 0+ and ≥1+ salmon at site d prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) were lower than predicted by the Habitat Quality Score (HQS) suggesting poorer populations than expected; the Habitat Utilisation Index (HUI) upper CLs were <1 for 0+ salmon in both scenarios and ≥1+ salmon following hydropower scheme construction therefore the observed populations were significantly lower than expected (Table 5.17). 0+ trout and ≥1+ trout (> 20 cm) observed densities prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) were lower than predicted by the HQS suggesting poorer populations than expected; the HUI upper CL was <1 for 0+ trout following hydropower scheme construction therefore this observed population was significantly lower than expected (Table 5.17). The observed densities of ≥1+ trout (<20 cm) prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) were higher than predicted (HQS) suggesting better populations than expected, but the HUI lower CLs were <1 therefore the observed populations were not significantly higher than expected (Table 5.17).

The observed density of 0+ salmon at site e prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) was lower than predicted by the HQS suggesting poorer populations than expected; the HUI upper CL was <1 therefore the observed populations were significantly lower than expected (Table 5.18). The observed densities of ≥1+ salmon prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) were higher than predicted (HQS) suggesting better populations than expected, but the HUI lower CLs were <1 therefore the observed populations were not significantly higher than expected (Table 5.18). The observed densities of all trout age/size groups prior to and following hydropower scheme construction (2007-2008 and 2010-2011 data) were lower than predicted by the HQS suggesting poorer populations than expected; the HUI upper CL was <1 for 0+ trout prior to hydropower scheme construction therefore this observed population was significantly lower than expected (Table 5.18).

Table 5.17 HABSCORE outputs for site d before (2007-2008) and after (2010-2011) the Allt Gleann Da-Eig hydropower scheme was constructed. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2007-2008	3.9	0.99	27.33	7.38	101.23	0.04	0.00	0.28	-3.21
2010-2011	0	0	70.67	17.64	283.19	0.00	0.00	0.04	
≥1+ salmon									
2007-2008	4.5	1.14	6.64	1.88	23.45	0.17	0.03	1.11	-1.77
2010-2011	0.10	0.31	6.62	1.85	23.74	0.05	0.01	0.31	-3.00
0+ trout									
2007-2008	6.8	1.73	4.93	1.26	19.36	0.35	0.05	2.42	-1.04
2010-2011	4.2	1.32	15.88	4.10	61.57	0.08	0.01	0.56	-2.53
≥1+ trout (<20 cm)									
2007-2008	15.0	3.82	2.30	0.51	10.37	1.66	0.27	10.42	0.50
2010-2011	17.5	5.45	4.48	0.99	20.38	1.22	0.19	7.67	0.20
≥1+ trout (>20 cm)									
2007-2008	2.4	0.63	0.61	0.18	2.03	1.02	0.31	3.42	0.01
2010-2011	0	0	0.42	0.13	1.41	0.74	0.22	2.45	-0.30

Table 5.18 HABSCORE outputs for site e before (2007-2008) and after (2010-2011) the Allt Gleann Da-Eig hydropower scheme was constructed. (Note: shaded area represents where the observed population was significantly higher (blue-HUI lower CL column) or lower (red-HUI upper CL column) than would be expected under pristine conditions).

Age/size group / fisheries data group	Observed number	Observed density	HQS (density)	HQS lower CL	HQS upper CL	HUI	HUI lower CL	HUI upper CL	Ln (HUI)
0+ salmon									
2007-2008	32.9	8.05	103.44	24.41	438.29	0.08	0.01	0.66	-2.52
2010-2011	39.1	12.07	54.24	13.41	219.43	0.02	0.03	0.85	-3.91
≥1+ salmon									
2007-2008	48.9	11.99	5.55	1.55	19.92	2.16	0.33	14.12	0.77
2010-2011	36.3	11.21	5.60	1.56	20.09	2.00	0.31	13.14	0.69
0+ trout									
2007-2008	2.4	0.60	9.86	2.52	38.63	0.06	0.01	0.42	-2.81
2010-2011	10.5	3.24	10.28	2.64	39.64	0.37	0.05	2.14	-0.99
≥1+ trout (<20 cm)									
2007-2008	2.0	0.49	2.24	0.50	10.03	0.22	0.03	1.42	-1.51
2010-2011	4.0	1.23	3.90	0.85	17.82	0.32	0.05	2.01	-1.14
≥1+ trout (>20 cm)									
2007-2008	0.0	0.00	0.24	0.07	0.77	1.03	0.32	3.37	0.02
2010-2011	0	0	0.35	0.10	1.17	0.88	0.26	2.97	-0.13

5.6.4 Salmonid growth rates

At site a, back-calculated length of trout at age 1 ranged between 54 and 64 mm; with the slowest growth in 2006 and fastest in 2010 (Table 5.19). Back-calculated length of trout at age 2 ranged between 95 and 120 mm, with the slowest growth in 2006 and fastest in 2009. At site b, back-calculated length of trout at age 1 ranged between 51 and 70 mm with the slowest growth in 2007 and fastest in 2010 (Table 5.19). Back-calculated length of trout at age 2 ranged between 115 and 122 mm, with the slowest growth in 2005 and fastest in 2008.

Table 5.19 Back-calculated lengths at age of trout in the upper and lower reaches of Allt Gleann Da-Eig. Note no 0+ trout were captured in 2011 at site b. Sites c and d were excluded from 2008 ageing data set as sites were combined and therefore not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))			
		1	2	3	4
a	2005	63±1(3)	114±11(3)		
	2006	54±9(2)	95±1(2)		
	2007	56±12(5)			
	2008	62± 3(5)	107±6(5)		
	2009	57± 6(15)	120±16(3)		
	2010	64±7(5)			
	*2011	57±4(4)			
b	2005	65±8(3)	115±18(3)		
	2006	57±10(8)	119±10(6)		
	2007	51± 4(4)			
	2008	68±3(4)	122±14(4)		
	2009	68± 4(12)	117±5(2)		
	2010	70±3(2)			
c	2003	85(1)	151(1)	210(1)	256(1)
	2004	61(1)	109(1)	152(1)	
	2005	60± 8(4)	112±2(4)		
	2006	66±8(4)			
	2008	63(1)	113(1)		
	2009	69±9(5)	134±15(2)		
	2010	75±11(3)			
d	2006	68±7(4)			
	2008	69±13(9)	126±15(3)	165(1)	
	2009	77±9(13)	137±15(3)		
	2010	72±8(10)			
	*2011	73(1)			
e	2009	69±8(8)			
	2010	82±12(2)			
	*2011	61±5(11)			

At site c, back-calculated length of trout at age 1 ranged between 60 and 85 mm with the slowest growth in 2005 and fastest in 2003 (Table 5.19). Back-calculated length of trout at age 2 ranged between 109 and 151 mm, with the slowest growth in 2004 and fastest in 2003. Back-calculated length of trout at age 3 ranged between 152 and 210 mm and at age 4 was 256 mm in (Table 5.19). At site d, back-calculated length of trout at age 1 ranged between 68 and 77 mm; with the slowest growth in 2006 and fastest in

2009 (Table 5.19). Back-calculated length of trout at age 2 ranged between 126 and 137 mm, and at age 3 was 165 mm. At site e, back-calculated length of trout at age 1 ranged between 69 and 82 mm, excluding 2011 data as this was based on observed mean length, and the slowest growth was in 2009 and fastest in 2010 (Table 5.19).

At site a, two salmon were captured in 2007 and back-calculated length of salmon was 51 mm at age 1, 86 mm at age 2 and 118 mm at age 3 (Table 5.20). At site c, back-calculated length of salmon was 42 mm at age 1, 106 mm at age 2 and 144 mm at age 3. At site d, back-calculated length of salmon at age 1 ranged between 46 and 54 mm and at age 2 was 82 mm (Table 5.20). At site e, back-calculated length of salmon at age 1 ranged between 59 and 61 mm, excluding 2011 data as this was based on observed mean length, with the slowest growth in 2010 and fastest in 2009 (Table 5.20).

Table 5.20 Back-calculated lengths at age of salmon in the upper and lower reaches of Allt Gleann Da-Eig. No 0+ salmon were caught in 2011 at sites a, c and d. Sites c and d are excluded from 2008 ageing data set as sites were combined and therefore not comparable. * see Section 3.3.4 for details.

Site code	Year class	Back-calculated length at age (mm ± S.D (n))		
		1	2	3
a	2004	51±9(3)	86± 5(3)	118±8(3)
c	2004	42(1)	106(1)	144(1)
d	2006	54±11(17)		
	2008	46(1)	82(1)	
e	2009	61± 8(28)		
	2010	59±8(13)		
	*2011	47±4(39)		

5.6.5 Eel

No eels were captured in the upper reaches of Allt Gleann Da-Eig throughout the study period. Numbers of eels captured at sites c-e varied between survey years and data suggest the lower reaches of Allt Gleann Da-Eig are used by small numbers of eels (Table 5.21). Despite variation in numbers of eels caught throughout the study period, there was no clear deviation of eel numbers during the hydropower scheme construction.

Table 5.21 Number of eel captured at survey sites in the lower reaches of Allt Gleann Da-Eig between 2007 and 2011. n/s = not surveyed.

Year	Site identifier		
	c	d	e
2007	0	0	1
2008	1	1	1
2009	n/s	n/s	n/s
2010	0	5	3
2011	1	1	1

5.6.6 Wetted Widths

A paired two sample t-test was performed on individual wetted widths (collected during HABSCORE) at site d; as this would have been located within the depleted reach had commissioning started. Essentially, the test was used to distinguish if there was any difference between wetted widths each time HABSCORE was performed to ensure comparison of density estimates over the study period. The 95% confidence limits (-1.585, 2.167) enclose zero with a p value of 0.750, n = 24; therefore there was no significant difference between wetted widths collected during the surveys pre and post construction, confirming comparability of density estimates.

5.7 Discussion

At Ardvorlich Burn, Douglas Water and Camserney Burn salmonid populations varied over the study period. Densities of fish varied both within and outside the impacted reach, thus natural fluctuations made it difficult to detect any impacts specifically associated with commissioning of the hydropower schemes. Furthermore, it should be noted that there are several limitations with the data sets within this chapter, including a lack of baseline and post-commissioning data and absence of control sites accounting for temporal and spatial variation. As such it has been difficult to draw definitive conclusions for each scheme documented in this chapter, as a lack of data further exacerbated the ability to detect any impacts.

In Ardvorlich Burn it was difficult to detect any impact of the scheme on fish density as the scheme commenced abstraction in 2011, only a few months before the 2011 surveys were conducted. Ultimately, any impact may not initially be detected from surveys carried out in the same year, especially as fish would not have spawned, or even undertaken spawning migrations since it was commissioned. 0+ fish data (pre and post-commissioning) are vital in order to allow the detection of any impact on spawning success and this cannot be done with the available data. Post-commissioning surveys would need to be carried out in September 2012 and ideally 2013 in order to assess if the scheme has had an impact on salmonid recruitment since abstraction began in 2011. Even so, with 2 years of post-commissioning data, it can be difficult to relate changes in fish densities specifically to the hydropower scheme as natural variation could mask the changes in response to abstraction. Even associating the fluctuations in density to natural variation cannot be definitively concluded without sufficient post-commissioning data.

Trout population densities declined at site D6 in the abstracted reach of Douglas Water and were lower in 2011 than in 2002, but this decline was also found at site D8, the control site upstream of the abstracted reach. At the other sites within the abstracted reach trout densities were higher in 2011 than 2002, but the time elapsed between the baseline surveys in 2002, commissioning of the scheme in 2008 and subsequent monitoring surveys 3 years later in 2011 makes identifying any impact due to flow regulations tremendously difficult. High inter-annual variability is a prominent feature of salmonids, as displayed in this thesis (Chapter 4) therefore surveys should be carried out in consecutive years before and after commissioning of the scheme to account for temporal variation.

Although baseline data pre hydropower commissioning were unavailable for Camserney Burn, densities of salmonids were compared between sites within and outside of the abstracted reach. Trends within trout and salmon densities observed at sites within the abstracted reach were mirrored at sites located outside the abstracted reach. Fluctuations in salmonid densities might therefore be suggested to be in response to natural variability as opposed to the commissioning of Camserney Burn. However, it is reinforced that natural fluctuations observed in the control reach may not necessarily confirm that there are not other impacts associated with the hydropower scheme; pre-commissioning data are needed to definitively confirm this which are not available. Pre-commissioning data are crucial to indicate the natural trends observed before abstraction, and ultimately support interpretation of any fluctuation in densities that are observed post abstraction.

In the River Callop, located in the Shiel catchment, 0+ trout and salmon densities declined at several sites in the depleted reach post hydropower commissioning. 0+ trout densities at site c3 located in the depleted reach, declined from the highest recorded in 2006 to zero in 2011. The River Callop run-of-river hydropower scheme, was commissioned in 2008, thus 0+ fish spawned in 2010 and 2011 post hydropower commissioning can be an indicator of any potential issues of the hydropower scheme on spawning success. A similar declining trend at site c3 was also found for ≥1+ trout densities between 2006 and 2011. The surveys carried out by MFC prior to commissioning did not involve any surveys in the reaches upstream of the then proposed intakes; all sites in the upper reaches in 2006 were within proposed abstracted reaches therefore there are no upstream control sites to compare data with for this study. While it should be noted that 0+ trout densities at the downstream control site, c1, also declined between 2006 and 2011 it should equally be noted that the upstream and downstream reaches of a river can be very different in their characteristics; hence the need for separate control sites in both the upstream and downstream reaches. The use of independent upstream and downstream control sites is vital when accounting for spatial variability, which is an important factor when determining if impacts are specifically in response to hydropower schemes. Ultimately, using control sites in the lower reaches as a control for sites in the upper reaches should be taken as indicative only and should not be used to conclude with confidence that changes in densities are most likely to be due to natural variability.

Nevertheless, at site c3 in the depleted reach, 0+ trout density estimates decreased from 7/100 m^2 in 2006 (prior to hydropower commissioning) to 1/100 m^2 in 2010. No 0+ trout were caught in 2011. Similarly at site c6 in the depleted reach, 0+ salmon density estimates decreased from 25/100 m^2 in 2006 to zero in both 2010 and 2011. A similar decreasing trend was also observed in 0+ salmon densities at sites c7 and c2, both located in the depleted reach. Again as there were no control sites or salmonid length data for surveys carried out prior to the scheme's commissioning, it is difficult to know whether these sites are naturally poor at supporting 0+ fish or whether they have declined in response to commissioning of the hydropower scheme. Growth of salmon and trout in the first year of life is an important indicator of survival of fish and is closely linked to abiotic and biotic factors; length data would have therefore provided information regarding the growth of salmonids in the upper reaches of the River Callop that could have been compared to those recorded after abstraction.

It is documented that a complex array of riverine habitats such as pools, riffles and backwater areas are preferred by salmonid fish (Matthews, 1985; Angermeier, 1987), but these habitats can be lost or damaged in depleted reaches, such as those experienced in run-of-river schemes. HABSCORE data were not collected prior to commissioning of the River Callop hydropower scheme therefore the data did not allow any assessment of potential impact of the hydropower scheme; the analysis was included however to identify the potential of each site for salmonid usage. Data suggest the habitat in the River Callop is suitable for supporting higher densities of 0+ salmonids than were found in the reach. Indeed, HABSCORE outputs at both c3 and c6 indicated that numbers of 0+ trout and salmon respectively, were significantly lower than expected, thus the reach has suitable habitat available to support much higher numbers of 0+ fish than were caught (according to HABSCORE). 0+ trout densities at site c6 were also significantly lower than expected. 0+ salmon were absent from site c3, presumably because this site is located in the upper reaches and thus inaccessible to adults. It should be noted, that rod catch data of adult return rates to the Shiel catchment indicated 32 adult sea trout returned in 2009 and 23 (17 sea trout and six finnock (sea trout that return the same year they smolted)) in 2010, perhaps giving reason for the decline observed in 0+ trout between 2010 and 2011. Nineteen salmon returned in 2009 and 36 salmon in 2010, it could perhaps be likely to therefore expect an increase in 0+ salmon in 2011 given the higher number of adult returns in 2010.

This was not observed and 0+ salmon remained absent, even though suitable habitat was available to support this age class. Perhaps adult salmon only occasionally penetrate this far upstream to spawn (given the higher numbers of ≥1+ salmon) but due to a lack of temporal data this cannot be definitively concluded. While rod catch data can be used to indicate the number of adults returning each year, it should be noted that an increase or decrease in these data can also be in response to changes in fishing effort or efficiency.

While there was a decline in 0+ trout at site c3 between 2006 and 2010 there was a simultaneous increase of ≥1+ trout between these years. This suggests that perhaps the site has become more suitable for the latter age class, due to a change of habitat, as a result of reduced flows. When low flows were simulated in an artificial stream, Davey *et al.* (2006) observed that two New Zealand fish species actively emigrated from riffles in direct response to reduced flows. Therefore, reductions in flow, which are undoubtedly experienced in abstracted reaches of run-of-river schemes, may have forced juvenile salmonids, which favour shallow riffle areas (Crisp, 2000), into deeper areas where they may face increased competition or predation from larger salmonids (North, 1979). Therefore, if site c3 has become more suitable for the larger fish as a consequence of habitat alteration, it is possible that ≥1+ trout have out competed 0+ fish resulting in a reduction of the latter but increase in the former age class. By contrast, Kubecka *et al.* (1997) reported that water abstraction in low-head hydropower schemes in the Czech Republic, resulting in low flows and thus depleted reaches, caused changes in species composition from large-bodied (adult brown trout) to small-bodied fish (juvenile trout). This said, it should be noted that generally ≥1+ trout populations have declined over the study period between 2006 and 2011 at all sites. Indeed, it has been documented that depleted flows, such as those that are possibly experienced in the depleted reach of the River Callop, have led to declines in populations of brown trout as a result of competition and predation (Jansson, 2002), partly as a result of reductions in macro-invertebrate abundances resulting from flow modification (Valentin *et al.,* 1996; Cereghino & Lavandier, 1998). Impairing invertebrate communities can greatly affect fish assemblages as they form the nutritional basis for many species (Fette *et al.,* 2007).

Ultimately, due to a lack of both temporal and spatial data it is difficult to conclude with any confidence whether the decline in salmon and trout populations in the abstracted reach of the River Callop is a natural population decline or impact of flow regulation. While it should be noted similar declines were observed in the 0+ salmon and trout populations at the downstream control sites it is highlighted upper and lower reaches of a river can be extremely different in characteristics. Therefore, it is not ideal to use downstream control sites as a control for sites in the upper reaches, and vice versa. A lack of temporal data further exacerbated the difficulty to allow confident detection of impacts specifically in response to the hydropower scheme.

Allt Gleann Da-Eig run-of-river hydropower scheme, located in the Tay catchment was commissioned post-2011 fisheries surveys; data were therefore analysed to assess any potential impact of the hydropower scheme's construction in 2010. There are many issues associated with the construction of intakes and outfalls, one being increased sedimentation which can have negative effects on the spawning grounds of salmonids. One of the key results from this scheme was the reduction of 0+ trout at sites b, c and d (all downstream of the intake) in 2011 (post construction). Acornley & Sear (1999) reported siltation of spawning gravels to be a major reason for the declines of salmonid stocks in the UK. Fine sediments tend to fill redds from the bottom up, reducing redd permeability and thus preventing aeration of the eggs and the removal of metabolites, which can lead to egg mortality (Barlaup *et al.,* 1994). Furthermore, silt and sand can potentially form a seal on the redd surface (Lisle, 1989) and reduce embryo survival. High levels of fine sediment can damage the embryos by abrasion (Jonsson & Jonsson, 2011) and form a barrier thus preventing the emergence of alevins (Crisp, 1993). Sear *et al.* (2008) further argued that accumulation of fine sediment within redds and/or the sedimentation of spawning gravels causes deleterious impacts on both incubating and emerging salmonids. It is therefore possible, that construction of the scheme increased sedimentation within the stretch of river downstream of the intake and reduced the quality of spawning habitat and/or impaired the incubation of eggs leading to a decline in 0+ fish. However, the number of adult returns to the Tay catchment should also be considered as this can have a direct effect on recruitment success (Crozier & Kennedy, 2001). In 2009, 288 adult sea trout (287 sea trout and one finnock) were recorded with a slight decrease in 2010 to 225 adult sea trout (222 sea trout and three finnock). It is therefore possible that the reduction in the number of adult returns (rod catch data) between 2009 and 2010 could account for the decrease in 0+ trout between 2010 and 2011. Impacts of construction are not necessarily limited to the depleted reach and could have potentially affected the site downstream of the outfall (by construction of the intake, pipeline and outfall). Densities at site e, downstream of the outfall however, were higher in 2011 (post construction) than 2007 (pre construction). As this was also noted at site a, upstream of the intake, increases in densities are suggested to be in response to natural variation.

5.8 Conclusion

During the current period of monitoring (i.e. prior to and following commissioning/construction of the hydropower schemes) it is considered there have been possible impacts of flow alteration by the River Callop hydropower scheme on salmon and trout densities at sites within the abstracted reach. However this interpretation should still be treated with caution due to the lack of control sites to definitively conclude this with confidence. During the current study period, it has been difficult to detect any impact of the Ardvorlich Burn, Douglas Water, Camserney Burn and Allt Gleann Da-Eig hydropower schemes on fish densities due to a lack of spatial and temporal data; data limitations have weakened data analysis and thus the confidence of conclusions drawn.

CHAPTER SIX

6 FRAMEWORK FOR FUTURE MONITORING OF FISH POPULATIONS USING BACI ANALYSIS

6.1 Introduction

The increase in the number of run-of-river hydropower schemes being developed in the UK has presented an array of conflicts and challenges. There is a requirement for increased use of hydropower, as a form of renewable energy, to reduce impacts that could lead to climate change. There are also legal obligations to prevent deterioration of the current services provided by water to people and the environment, including no deterioration of existing water quality implemented by the Water Framework Directive (WFD). Potential changes to fish populations, caused by run-of-river hydropower schemes (Chapter 2) could impact upon the UK statutory bodies' obligations to deliver a range of duties; and indeed the requirements of the WFD. The WFD currently aims to achieve at least "good ecological status" (GES) or, for water bodies designated as artificial or heavily modified, "good ecological potential" (GEP) in all surface water bodies by 2015. Thus there is a requirement to prevent deterioration in the status or potential of any water quality element. It is therefore important that there is a full understanding of the extent of environmental protection that is necessary to allow the objectives that these obligations require to be achieved (Crocker, 2010). As such, it is critical that hydropower developments are undertaken within the criteria of environmental legislation (Aldrick, 2010), especially those of the WFD.

Given the objectives of the WFD, it is vital that any net change in the water environment can be measured; such as changes in the status of fish populations. This may be achieved by carrying out long-term monitoring surveys both before and after commissioning of a run-of-river hydropower scheme. Although the present study was across a relatively short time scale, due to various financial and development constraints, this research demonstrated that monitoring programmes can be used to detect the impact of run-of-river hydropower schemes on fish populations (Chapters 4 & 5). Findings from some of the study schemes however, should be treated with caution due to inconsistent monitoring by some hydropower developers outside the control of this project resulting in lack of comparable data. It is proposed on this basis, that long-term monitoring, including long-term baseline data may be required (Chapter 5); however collection of long-term baseline data may present financial and planning difficulties for hydropower developers. This chapter discusses the importance of implementing a standard monitoring programme and aims to derive an optimal monitoring design (assuming there are no funding/time constraints as with this research) for run-of-river hydropower schemes for robust impact assessments to be performed; this allows for any significant changes within fish populations to be detected. A Before After Control Impact (BACI) analysis was used to achieve this goal, which compares mean density of fish before and after the commissioning of a hydropower scheme in control and impacted reaches. Based on data acquired from this investigation, BACI analysis estimated resource requirements in terms of the number and frequency of sites necessary to sample to achieve predetermined levels of discrimination (Sedgwick, 2006); where data from this project conformed to the requirements of BACI a full impact assessment was undertaken. Impact assessment in this chapter is defined as the quantitative detection and measurement of change within salmonid populations.

6.2 Environmental effects monitoring

Environmental monitoring programmes can take many forms, but in the context of this study refers to impact assessment monitoring. This form is targeted at assessing human impacts on the natural environment (Downes *et al.,* 2002); in this study the impact of run-of-river hydropower schemes on fish populations.

Environmental effects monitoring is required when the objectives of a monitoring programme require the actual effects to be determined. In terms of an impact assessment, the key objective is to detect a change in a given variable, such as fish populations. Monitoring a given variable allows detailed scientific information to be gained and provides not only information about the current status of the environment but also feedback about the actual environmental impacts of a project, such as hydropower generation. Monitoring ultimately provides a better understanding of the cause and effect relationships (Roni, 2005) of a given action but equally reduces any uncertainty surrounding the effects of management actions on population dynamics of target species, such as Atlantic salmon. Essentially monitoring enhances the predictive capability for assessing potential effects (Lohani *et al.*, 1997). Assessing the effectiveness of mitigation measures (if applied) allows any problems to be recognised and for modifications to be made where required.

Lohani *et al.* (1997) defined environmental effects monitoring as *"the repetitive and systematic measurement of the characteristics of environmental components to test specific hypotheses of the effects of human activity on the environment. Environmental monitoring is undertaken primarily to determine the environmental effects of human activities and secondarily to increase understanding of cause-effect relationships between human activity and environmental change".*

The implications of this definition are that:

1. Environmental monitoring programmes should involve repetitive sampling over a number of years;
2. Environmental monitoring programmes should be scientifically rigorous and be based on testable hypotheses;
3. Sampling programmes designed to test the hypotheses should be such that the results may be used to detect temporal trends and/or spatial differences; and
4. Environmental monitoring programmes should attempt to establish empirical links between human activities and their effects on the environment.

Without environmental monitoring, it is not possible to determine whether the environment is being protected sufficiently as there is no mechanism for evaluating the success of mitigation measures undertaken, thus any impacts caused could continue indefinitely (Lohani *et al.*, 1997). Critically, environmental monitoring is absolutely essential if environmental degradation is to be controlled. By generating quantitative information, it is of great importance to those that plan developments and manage environmental resources as it provides the backbone upon which to base their decisions and follow-up activities (Everitt, 1992).

In general, environmental monitoring programmes will collect data for one or more of the following purposes (Everitt, 1992):

1. To establish a baseline; that is, gathering information on the basic site characteristics prior to development or to establish current conditions;
2. To establish long-term trends in natural unperturbed systems to establish natural baselines;
3. To estimate inherent variation within the environment, which can be compared with the variation observed in another specific area;
4. To make comparisons between different situations (for example, pre-development and post-development; upstream and downstream; at different distances from a source) to detect changes; and
5. To make comparisons against a standard or target level.

The ten run-of-river hydropower schemes surveyed in this study were monitored to collect data for the five purposes listed above. As there is currently no legislation in Scotland stating the amount of monitoring that must be implemented pre- and -post-commissioning of a hydropower scheme, there were differences in the amount of monitoring implemented for each of the schemes. Chapters 4 and 5 highlighted the difficulties that can arise when sufficient data were not collected; baseline data gathered for each of the schemes in this investigation differed considerably, which ultimately affected the strength of the data analyses and the conclusions that were drawn. It is for this reason that a generic monitoring programme should be produced and implemented for all future run-of-river hydropower schemes.

6.2.1 Current monitoring of hydropower schemes

According to the consultation that the EA released following the Good Practice Guidelines to the Environment Agency handbook (EA, 2009b) if developers were to follow the revised Good Practice Guidelines, hydropower developers would not generally be required to monitor the potential environmental effects of their schemes (EA, 2009b). The only time that monitoring may be required is if the hydropower scheme is located in a sensitive area such as a SAC. Where there is uncertainty that the guidelines will afford adequate protection, pre-and/or post-determination monitoring may be required. Under these circumstances, although it is suggested that the timescales need to be sufficient to allow for natural variation in conditions over time, monitoring will not generally be open ended (EA, 2009b). The Scottish Executive Fisheries Committee (SEFC) however (now absorbed into SEPA) recommend that "following completion of the scheme, quantitative sampling of the fish populations at the census sites should continue at intervals of two or three years for a period of 10-12 years" (Robson *et al.,* 2012). Nevertheless, historically operational licences have been granted without any substantial pre-or post-monitoring of the impact of run-of-river hydropower developments on the ecology and other functions of rivers. Additionally, it is suggested that data collected for environmental impact assessments are often of such short duration and poor quality that the possibility of either detecting or quantifying impacts if they occur is precluded (Warnken & Buckley, 1998).

However, driven by the Habitats Directive, there is a requirement to promote the maintenance of biodiversity by maintaining/restoring natural habitats and wild species listed in the Annexes to the Directive at a favourable conservation status. This includes the introduction of robust protection for those habitats and species of European importance. The Habitats Directive thus created a network of protected areas around the European Union of national and international importance, such as SAC's, referred to as Natura 2000 sites. According to Article 3 (3) of the Habitats Directive - "Member States shall endeavour to improve the ecological coherence of Natura 2000 by maintaining, and where appropriate developing, features of the landscape which are of major importance for wild fauna and flora". Additionally, Regulation 48 of the Habitats Regulations 1994 governs the assessment of implications for a European site.

Assessments are therefore required when a plan would be likely to have a significant effect on a European wildlife site. It is vital that information of this nature is highlighted to potential developers.

A much greater level of guidance should be available to hydropower developers, which emphasises the need for well-defined monitoring programmes. Although a number of environmental operating conditions are set upon granting of scheme approval, implemented through permits and licences, and SEPA guidance indicates what baseline data must be collated prior to applying for a hydropower scheme (CAR, 2011), there are still no definitive monitoring programmes allocated to the commissioning of run-of-river hydropower schemes. Whilst the EA Good Practice Guidelines (EA, 2009a) and SEPA Guidance on Run-of-River Hydropower Schemes (SEPA, 2010a) provide summary guidance on requirements for developers of new run-of-river hydropower schemes, these provide no definitive monitoring protocol for undertaking EIAs prior to the application, including location, frequency and duration of surveying; despite being a pre-requisite of the CAR licence. Similarly, no definitive monitoring protocol for post-project monitoring of a scheme is provided, including type of surveys and for what purpose, frequency and duration of surveying. Developers may not be aware of data requirements in terms of number of control and impact sites and baseline data, resulting in many schemes monitored inconsistently or having no post-operational monitoring.

Chapter 5 highlighted limitations of data collected for several schemes, namely Douglas Water, Ardvorlich Burn and the River Callop. Much of the monitoring for these schemes was undertaken outside of this project by other organisations and therefore the monitoring programmes for these schemes were different to that of those undertaken for this research; causing several constraints during data analysis and impact assessments. This reinforces the need for robust monitoring designs to be implemented that incorporate the necessary requirements for full analysis to be conducted. Unfortunately, due to the resources necessary to do so, there is an extreme paucity in studies of this kind as ecology monitoring has not been adequately funded, designed, implemented or even reported (Roni, 2005) in many cases. Given the high level of spatial and temporal heterogeneity of streams (Downes *et al.,* 2002), detecting small changes in biota and particularly fish response to habitat alteration can be extremely challenging (Minns *et al*., 1996). As there is such uncertainty however, surrounding the potential implications of run-of-river hydropower schemes, it is suggested that monitoring be considered an integral part of the project cycle if improvements are to be seen from one project to the next. It is the key to increasing our scientific understanding of environmental impacts.

6.3 BACI design

Reeves *et al.* (1995) suggested that effective monitoring requires an understanding of three components; the temporal and spatial scales, the nature of both the impact and the response, and historic and current conditions. These factors are especially important when monitoring fish populations that display great spatial and temporal natural variability (Chapters 4 & 5). A common approach for determining biological response to habitat alteration is to measure conditions before and after a treatment; in this instance changes in fish density before and after hydropower commissioning. This method of analysis is referred to as a BA (Before After) approach, and therefore generally is replicated in time (temporal) rather than space (spatial) (Roni, 2005). Lichatowhich & Cramer (1979) examined several abundance, survival and life history parameters for salmonid fishes and suggested that BA studies of abundance may require 20-30 years to produce an 80% chance of detecting a change of 50% or more. This is an incredibly long time period that in reality would not be appropriate to propose to future hydropower developers. It is suggested however, that the minimum requirements for an effective monitoring programme in terms of impact assessment are having sufficient samples from both impact and control locations during both the before and after periods (Downes *et al.,* 2002). The BACI approach, which is replicated both temporally and spatially (Roni *et al.,* 2003), covers all these requirements and is therefore a suggested improvement on the BA design; this is a classic method for measuring the potential impact of a discharge, disturbance, or event on the fish and invertebrates of a stream.

The BACI approach is considered to be the best current design for impact assessment (Roni *et al.,* 2003) as there are essentially two treatments; before-after, which is of primary interest, and control-impact, which is of secondary interest. In this design the selection of appropriate controls is essential in which a control site is evaluated over the same time period as the treatment (impact) site. The addition of control sites account for environmental variability and temporal trends (a trait common of salmonid populations) found in both the control and impact areas, and subsequently increases the ability to differentiate the effects of a particular treatment from natural variability (Smith *et al.,* 1993), i.e. change in impact area – change in control area. This research has indeed reflected the variable and temporal trends salmonid populations exploit (Chapters 4 and 5), specifically the temporal variation observed in baseline data (pre-commissioning) of 0+ salmon at Keltney Burn (Section 4.3); thus a monitoring programme accounting for temporal and spatial variance is vital when assessing potential impacts of fish populations.

A replication BACI design i.e. monitoring multiple controls and impacts before and after is potentially the most powerful of all study designs (Roni, 2005). As the approach includes replication in both time and space, both spatial and temporal replications are accounted for increasing statistical power further to detect treatment effects from natural variability; this is critically important in studies regarding salmonid populations. This type of sampling strategy however is more challenging and costly to implement (Downes *et al*., 2002). Whilst many authors feel that the BACI approach is a much needed improvement on the BA design, the choice of a control site is crucial, with evidence suggesting a BACI design with a poorly chosen control site can be less powerful than the uncontrolled BA design (Roni *et al*., 2003). While no one design is correct for all situations, Roni (2005) suggested that the ideal BACI design includes many paired treatments and controls across the rivers that are monitored for many years. This is perhaps the optimum design given no limitations in funding and while this may be challenging to design and implement (Downes *et al.,* 2002) it is the method of monitoring that should be aimed for to quantify the impact of run-of-river hydropower schemes on fish populations.

6.3.1 Post treatment

Whilst it is highly recommended to use BACI studies as the method for assessing biological response to habitat alterations, in some instances this cannot be done; if it is an existing hydropower scheme it would not be possible to measure the before conditions (assuming pre-commissioning data were unavailable for the scheme in question). Under these circumstances, the treated reach is compared to areas thought to be similar in the absence of the activity in the form or a control or reference site (section Controls and references); this is referred to as a post-treatment design. Essentially, post-treatment designs are retrospective studies, replicated spatially rather than temporally. While the BACI approach is unsuitable for schemes that are already operational with no baseline data available, a two-tier approach could indeed be proposed by using the post-treatment method for schemes that began operation without pre-commissioning data. It should be noted however that lack of data gathered prior to a treatment (impact) is considered to be one of the most common deficiencies in impact monitoring (Downes *et al.,* 2002) and therefore post-treatment should only be considered secondary to full BACI analysis.

6.3.2 Controls and references

The importance of control and reference sites cannot be understated in the context of impact assessment (Roni, 2005). When a common trend occurs in the control/reference and treatment sites, the control site provides a basis of comparison and thus prevents the trend from being interpreted as a response to the treatment, such as a reduction in flow. It essentially serves as a covariate to account for natural variability and accounts for a portion of the natural background variation that may mask detection of a true response to the treatment, i.e. flow alteration. Although the purpose of a control and reference site is identical, there is a slight distinction between the two. A control is generally defined as being identical to the impact (treatment) site with the exception of the treatment (Downes *et al*., 2002); in the case of hydropower generation the abstraction of water. It has been discussed, however, that no true controls exist in field studies because no two sites, reaches, watersheds, or estuaries are identical (Roni *et al.,* 2003); indeed it would be tremendously difficult to find a control site that is identical in habitat to that of the impact site, which leads to the term reference site. A reference is defined as the ideal or pristine state, with conditions unaltered by human activities (Downes *et al*., 2002) or representing a range of pre-disturbance conditions (Reeves *et al*., 1995). In either case, controls or reference sites should be as close as possible to an independent replicate of the treatment and therefore should reflect similar aspects in terms of land use, geology, hydrology, biology and other physical features (Roni, 2005).

6.4 BACI analysis

As discussed, the purpose of an impact assessment in the context of this study is to provide statistically robust evidence that a meaningful change in fish density has occurred. However, proving a change in density was actually caused by the pressure under observation (hydropower commissioning) and not attributed to coincidental temporal and spatial influences can be challenging; especially given the natural fluctuations salmonid populations exploit. A study design must therefore account for extraneous influences by including density data from the same sites before the impact to eliminate spatial differences and through comparison of a reference (un-impacted) site to eliminate natural temporal variations. Sedgwick (2006) documented the procedure to apply these principles to analyse fish population changes in space and time to determine resource requirements and ultimately perform impact assessments. The assumptions and equations documented therein were employed in this investigation.

The BACI analysis was performed on data from the lower reaches of Keltney Burn; this scheme was the most ideal in terms of meeting the requirements of the BACI design with more than one impact site over several years pre and post treatment (hydropower commissioning). The BACI analysis was performed using data from this study in two different combinations (varying in the number of years sampled and number of control sites) to show the effects of temporal and spatial variance and the downfalls of limited years or sites. Control rivers and sites were chosen based on their similarity to Keltney Burn. Rivers located in the same catchment (Tay) as Keltney Burn and sites in the lower reaches were selected. In both combinations the analysis was run using 0+ and ≥1+ age groups of salmon and trout; age groups and species were treated separately. Combination one used data from two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (seven years) (Table 6.1). Combination two used data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (four years). Density data from two impact sites at Keltney Burn (sites d and e) were used in the analysis for both combinations. It should be noted that the BACI analysis was only performed using sites in the lower reaches of rivers as the density of trout in the upper reaches was often very low in comparison and salmon were usually absent.

Table 6.1 Schematic example of BACI quadrant based on number of years and sites in combination one (two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (seven years)). x represents density (N/100 m^2) of fish. Age groups (0+ and ≥1+) of each species (salmon and brown trout) are treated separately.

		BEFORE						AFTER		
		2005	2006	2007	2008	2009		2010	2011	
CONTROL	Site 2f	x	x	x	x	x		x	x	
	Site 3d	x	x	x	x	x		x	x	
							Mean			Mean
IMPACT	Site 2d	x	x	x	x	x		x	x	
	Site 2e	x	x	x	x	x		x	x	
							Mean			Mean

Essentially, mean densities of fish are compared before and after hydropower commissioning at control and impact sites; the mean density value in the impact/after box (red text) (Table 6.1) ultimately isolates any impact due to a treatment (hydropower commissioning) from natural variability.

In all instances, the data were checked for normality of error variance using a one-sample Kolmogorov-Smirnov test and for homogeneity of variance using the Bartlett's tests (Dytham, 2009), and transformed (LN or LOG) where necessary. It should be noted that while transforming only some of the data is perhaps not ideal, data had to be manipulated when necessary to meet the BACI assumptions (Sedgwick, 2006). It is important to note however that each data set for each species and age group were analysed in isolation and therefore there was at no point a comparison between transformed and raw data. Listed below are the transformation of data to conform to assumptions of normality and homogeneity of variance.

Combination 1 0+ trout = LOG transformed density estimates (N/100 m^2)
Combination 1 ≥1+ trout = LN transformed density estimates (N/100 m^2)

Combination 2 0+ trout = LN transformed density estimates (N/100 m^2)
Combination 2 ≥1+ trout = LOG transformed density estimates (N/100 m^2)

Combination 1 0+ salmon = LN transformed density estimates (N/100 m^2)
Combination 1 ≥1+ salmon = Raw density estimates (N/100 m^2)

Combination 2 0+ salmon = Raw density estimates (N/100 m^2)
Combination 2 ≥1+ salmon = Raw density estimates (N/100 m^2)

Impact assessment calculations identified the mean change in fish density and whether a significant effect occurred by calculating confidence limits of the change (*t*-statistic; $P < 0.05$) (see Section 6.4.1.2), once a resource calculation has been performed on the data. Resource equations traditionally use a small pilot study or routine monitoring programme data from similar study rivers to determine the level of future sampling. In the context of this thesis, resource equations derived from data collected in this study (see Section 6.4.1.1), were used to establish if sufficient sites were sampled enough to identify a population change (i.e. the precision level) within a stated level of probability (e.g. 0.8 or 80%) and statistical power (e.g. 0.05 or 5%). It is important to consider the precision level that must be achieved. In this context, precision is associated with the "noise" (expressed as the variance) generated by the spatial and temporal variations in fish populations, and is usually reduced by larger sample sizes or repetitive surveys. A reliable estimate will have a low variance. The precision level deemed biologically meaningful and within the realms of feasible resource allocation was 50% of the mean pre-impact density (Cowx, 1996).

Specifically, the following steps were followed:

1. Mean density was calculated for 0+ and ≥1+ age groups of trout and salmon in the Keltney Burn impact reach before the hydropower development was commissioned.
2. The target variance (Sedgwick, 2006) for a 50% change in the mean pre-impact density for 0+ and ≥1+ age groups of trout and salmon was calculated:

   (50% of mean density before/(Ø*SQRT(2)))^2 Equation 6.1
   Ø is a given value relating to the associated degrees of freedom determined by: (number of control sites + number of impact sites) - 2

3. The actual variance (V(x)) (Sedgwick, 2006); Equation 6.2) of the full BACI quadrant for 0+ and ≥1+ age groups of trout and salmon was calculated.

V(x) = (Vytr)*(1/(mB*nT)+1/(mA*nT)+1/(mB*nC)+1/(mAnC))   Equation 6.2

Vytr = Residual variance (Error Mean Square (EMS) of a two-factor ANOVA without replication)

mA = No. of occasions after the event (years)

mB = No. of occasions before the event (years)

nT = No. of test (i.e. impact) sites

nC = No. of control (i.e. control) sites

4. The actual variance was compared to the target variance to identify if there were sufficient data to allow a significant impact to be detected (Table 6.2).
5. If the actual variance was greater than the target variance a statistically significant 50% change in the mean pre-impact density could not be identified. In this instance, the data were treated as a pilot study and a resource calculation was performed, i.e. the number of years and sites in Equation 6.2 was increased to establish how many sites and years of data would be required to derive statistically robust outputs. Please note, the resource calculation is based on the assumption that spatial and temporal variation would persist if more sites and years were sampled.
6. If the actual variance was less than the target variance a statistically significant 50% change in the mean pre-impact density could be identified and the impact assessment was performed (Equation 6.3). The impact is calculated from the differences in mean abundance derived from the four segments of the BACI design. This is defined as:

(Change in impact (or test) area) – (Change in control area).

i.e. Impact (**x**) = (y AT - y BT) – (y AC - y BC)   Equation 6.3

y BT is the mean abundance, over all sites and times, **before** the event, in the **test** area.

y AT is the mean abundance, over all sites and times, **after** the event, in the **test** area.

y BC is the mean abundance, over all sites and times, **before** the event, in the control area.

y AC is the mean abundance, over all sites and times, **after** the event, in the control area

General Linear Models (A four-factor, spatially and temporally nested ANOVA model) (GLM; type III sums of squares) performed in Minitab 16 were then produced to enable greater discrimination of the variance sources and therefore the causal influences of the population change.

Table 6.2 Actual variance and target variance for 0+ and ≥1+ brown trout and salmon for combination one (data from two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (seven years)) and combination two (data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (four years)). Bold text indicates when actual variance is below target variance allowing a full impact assessment to be conducted.

	Combination one		Combination two	
	Target variance	Actual variance	Target variance	Actual variance
0+ Brown trout	0.005	0.014	0.036	0.119
≥1+ Brown trout	0.015	0.036	0.006	0.018
0+ Salmon	**0.326**	**0.089**	41.853	130.260
≥1+ Salmon	6.718	29.207	16.001	41.048

6.4.1 Results

According to the outputs of the resource calculation, the minimum number of sites and years that need to be monitored to statistically detect a 50% change in 0+ and ≥1+ trout and salmon densities at Keltney Burn are presented in Section 6.4.1.1 (Tables 6.3-6.9), including the overall resource commitment, i.e. total number of sites sampled over the study period. Generally, if more sites are sampled in each study year the monitoring can be over a shorter period and *vice versa*. Red text in Tables 6.3-6.9 denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. For example, while the target variance is reached by sampling four control and four impact sites, 6 years before and 6 years after a scheme is commissioned, it is also achieved by sampling four impact and four control sites only 2 years before and 2 years after a scheme is commissioned (Table 6.3); thus less resource commitment is needed in the latter. Red text in bold (Tables 6.3-6.9) indicates the variance associated with the monitoring design requiring the fewest surveys (numbers in brackets) to achieve a variance lower than the target variance. Ultimately, indicating the most cost-effective monitoring design in order to acquire enough data to allow a statistically robust impact assessment to be performed.

6.4.1.1 Resource calculations

Combination one **2 control sites** - Keltney Burn (site f) and Camserney Burn (site d)
2 impact sites - Keltney Burn (sites d & e)
5 years' pre data - (2005-2009), **2 years' post data** (2010-2011)

Table 6.3 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in 0+ trout density at Keltney Burn; combination one (data from two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (7 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target variance	Actual variance for a specified number of years				
		2	3	4	5	6
2	0.005	0.020	0.014	0.010	0.008	0.007
3	0.012	0.014	0.009(36)	0.007(48)	0.005(60)	0.005(72)
4	0.016	**0.010**(32)	0.007	0.005	0.004	0.003

Table 6.4 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in ≥1+ trout density at Keltney Burn; combination one (data from two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (7 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target variance	Actual variance for a specified number of years					
		2	3	4	5	6	7
2	0.0154	0.051	0.034	0.026	0.021	0.017	0.0147(56)
3	0.037	**0.034**(24)	0.023(36)	0.017(48)	0.014(60)	0.011(72)	0.010
4	0.048	0.026	0.017	0.013	0.010	0.009	0.007

Determined by the outputs of the resource calculation i.e. when the actual variance was below the target variance, a full impact assessment could be conducted on 0+ salmon, using combination one (Table 6.2). See Section 6.4.1.2 for further details and results on the impact assessment of 0+ salmon.

Table 6.5 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in ≥1+ salmon density at Keltney Burn; combination one (data from two control sites (Keltney Burn site f and Camserney Burn site d) in 2005-2011 (7 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target Variance	Actual variance for a specified number of years					
		2	3	4	5	6	7
2	6.718	41.724	27.816	20.862	16.690	13.908	11.921
3	16.142	27.816	18.544	13.908(48)	11.126(60)	9.272(72)	7.947(84)
4	20.775	20.862	13.908(48)	10.431	8.345	6.954	5.961
5	23.393	**16.690**(40)	11.126	8.345	6.676	5.563	4.768

Combination two **5 control sites** - Keltney Burn (site f), Camserney Burn (site d), Innerhadden Burn (site f), Inverhaggernie Burn (site d), and Allt Gleann Da- Eig (site e)

2 impact sites - Keltney Burn (sites d & e)

2 years' pre data - (2007-2008), **2 years' post data** (2010-2011)

Table 6.6 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in 0+ trout density at Keltney Burn; combination two (data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (4 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target variance	Actual variance for a specified number of years				
		2	3	4	5	6
2	0.013	0.170	0.113	0.085	0.068	0.057
3	0.031	0.113	0.075	0.057	0.045	0.038
4	0.040	0.085	0.057	0.042	0.034(80)	0.028(96)
5	0.045	0.068	0.045	0.034(80)	0.027	0.023
6	0.048	0.057	0.038(72)	0.028	0.023	0.019
7	0.050	**0.049**(56)	0.032	0.024	0.019	0.016

Table 6.7 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in ≥1+ trout density at Keltney Burn; combination two (data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (4 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target variance	Actual variance for a specified number of years				
		2	3	4	5	6
2	0.002	0.025	0.017	0.013	0.010	0.008
3	0.005	0.017	0.011	0.008	0.007	0.006
4	0.006	0.013	0.008	0.006	0.005(80)	0.004(96)
5	0.007	0.010	0.007	0.005(80)	0.004	0.003
6	0.008	0.008	0.006(72)	0.004	0.003	0.003
7	0.008	**0.007**(56)	0.005	0.004	0.003	0.002

Table 6.8 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in 0+ salmon density at Keltney Burn; combination two (data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (4 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target Variance	Actual variance for a specified number of years				
		2	3	4	5	6
2	14.941	186.086	124.057	93.043	74.434	62.029
3	35.900	124.057	82.705	62.029	49.623	41.352
4	46.202	93.043	62.029	46.521	37.217(80)	31.014(96)
5	52.025	74.434	49.623(60)	37.217(80)	29.774	24.811
6	55.731	62.029	41.352	31.014	24.811	20.676
7	58.266	**53.167**(56)	35.445	26.584	21.267	17.722

Table 6.9 The number of control and impact sites to be sampled, the associated target variance and the number of years of sampling required to achieve the actual variance to statistically detect a 50% change in ≥1+ salmon density at Keltney Burn; combination two (data from five control sites (Keltney Burn site f, Camserney Burn site d, Innerhadden Burn site f, Inverhaggernie Burn site d and Allt Gleann Da-Eig site e), in 2007-2008 and 2010-2011 (4 years)). Red text denotes the variance associated with the fewest number of sites that must be surveyed, per number of years, for the actual variance to be lower than the target variance. Numbers in brackets indicate the total number of sites sampled over the time period.

Number of control & impact sites	Target variance	Actual variance for a specified number of years				
		2	3	4	5	6
2	5.712	58.640	39.093	29.320	23.456	19.547
3	13.725	39.093	26.062	19.547	15.637	13.031(72)
4	17.664	29.320	19.547	14.660(64)	11.728(80)	9.773
5	19.891	23.456	15.637(60)	11.728	9.382	7.819
6	21.308	**19.547**(48)	13.031	9.773	7.819	6.516

It is apparent from the analysis of the two combinations that it is much more advantageous to identify a local reference reach with several years of pre- and post-commissioning data (combination one) (Tables 6.3-6.5) than to have several reference reaches and only 2 years' pre- and post-commissioning data (combination two) (Tables 6.6-6.9). This is because much less resource commitment (number in brackets) was needed for the target variance to be achieved. Using the outputs from 0+ trout data as an example, the least amount of resource commitment needed for the target variance to be achieved in combination one was by surveying a total of 32 sites (four control/impact sites, 2 years before/after a scheme is commissioned) (Table 6.3). By comparison, the least amount of resource commitment needed in order for the target variance to be achieved using combination two was a total of 56 sites (seven control/impact sites, 2 years before/after hydropower commissioning) (Table 6.6). Ultimately, for each species and age group, less resource commitment was needed for the target variance to be achieved in combination one than combination two. Consequently, recommendations for monitoring programmes are based on the resource calculations produced from combination one.

From an overall resource commitment perspective, it is suggested that at least two years' monitoring pre- and post-commissioning monitoring of a hydropower scheme is implemented. As the outputs of Tables 6.3-6.5 differed in the number of sites required for the actual variance to be below the target variance (four sites (Table 6.3); three sites (Table 6.4); five sites (Table 6.5)), the precautionary approach was adopted and thus the highest number of sites needed is proposed; this is to ensure with confidence that the target variance is met. Five sites should therefore be sampled in the control and impact reach. However, given the large temporal variability in salmonid populations (Chapters 4 and 5) **it is recommended that sites are sampled for a minimum of 3 years before and after the commissioning of a hydropower scheme. Based on this recommendation, according to the resource calculation, four sites should be sampled in the impacted reach along with four reference sites.** This was determined by the same approach as mentioned above and using the highest number of sites that was needed for the target variance to be achieved when sampling 3 years before/after a scheme is commissioned, from the three outputs of combination one (Tables 6.3-6.5). Additionally, it is recommended that the resource calculation should be calculated after the first 3 years of monitoring to confirm that enough data have been collected to allow the BACI analysis to be performed successfully at a later stage post commissioning.

6.4.1.2 Full impact assessment for 0+ salmon

A full BACI design was used to assess the effects of the Keltney Burn run-of-river hydropower scheme on 0+ salmon densities in the Keltney Burn, using Camserney Burn as a control (combination one). The mean change in 0+ salmon density in the lower reaches of Keltney Burn after flow modification was -0.24 ± 2.44 fish/100 m^2.

A four-factor, spatially and temporally nested ANOVA model (GLM; type III sums of squares) was constructed to demonstrate how the sources of variance contributed to the eventual outcome. Period (before versus after the flow modification) and river (impact (Keltney Burn sites d and e) versus control (Keltney Burn site f and Camserney Burn site d) were fixed factors. Random factors were year nested within period and site nested within river. A term that accounted for the interaction between the main effects of period and river enabled detection of an effect of flow modification on 0+ salmon density. In this instance, the interaction was not significant (F = 1.08, P = 0.313; Table 6.10), i.e. there was no influence of the flow modification on 0+ salmon density. Therefore, **the null hypothesis was accepted, i.e. there are no significant differences in the 0+ salmon density in Keltney Burn before and after the changes in flow regime that can be attributed to the commissioning of the hydropower scheme. However, this result should be treated with caution as the control site (Keltney Burn f) was compromised by the relocation of the outfall.** See Section 6.4.2.

Table 6.10 General Linear Model (four-factor, spatially and temporally nested ANOVA model) of natural log (LN+1) transformed 0+ salmon density in the Keltney Burn (impact) and Camserney Burn (control) versus river, period, sites and years.

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Sites(River)	2	2.4892	2.4892	1.2446	12.08	0.001
Years(Period)	5	2.7331	2.7331	0.5466	5.30	0.004
River	1	0.3961	0.1811	0.1811	0.17	0.715
Period	1	0.0805	0.0805	0.0805	0.15	0.717
River*Period	1	0.1114	0.1114	0.1114	1.08	0.313
Error	17	1.7520	1.7520	0.1031		
Total	27	7.5623				

6.4.2 Limitations with BACI analysis

BACI analysis has been used in this investigation to show how to execute an impact assessment when sufficient monitoring has been performed; it also provides a framework of what is expected in the future for effective and robust monitoring designs to be achieved. It must be noted however that there are limitations within this data set. One of the fundamental features of performing a BACI analysis is ensuring there are adequate numbers of impact sites and control/reference sites to enable isolation of natural variance from the potential impact itself, this is why designing a monitoring programme is crucial. Within this data set however it must be made clear that the Keltney Burn control site (f) is located within/adjacent to the outfall. The use of this site within the analysis could therefore be criticised regarding its representation of a control site as a small proportion of the site is located within the depleted reach. This site was originally chosen as it was downstream of the proposed outfall; however, the location of the outfall was changed at a later stage before commissioning of the scheme commenced, however, several years of surveys had already been carried out at this site before the location was changed. In spite of this, the site was kept at the same location due to its quality and representation of several habitat features; furthermore it provides comparable data sets. It should also be noted that the outfall is only just upstream of the confluence of Keltney Burn with the River Lyon, thus it was not possible to have a site any further downstream than the current position of site f.

Additionally, control sites from different rivers within this study were used as control sites for Keltney Burn. Although all these sites were located in the lower reaches of rivers and within the same catchment as Keltney Burn, their use within this analysis could be perceived as a weakness given that these sites were not located within the same reach and thus not a true control. Nevertheless, it has been documented that a similar near-by river can provide a comparable reach but the risk of including the effect of other variables is increased (Sedgwick, 2006).

6.5 Conclusions

At the moment monitoring programmes do not exist for any scheme designs and therefore an extensive programme of monitoring to reduce the uncertainty surrounding hydropower is urged. Not only will monitoring allow data collection to assess potential impacts but it will further ensure that schemes are operating within the terms of their licence, if not it gives opportunity for remedial action to be taken where necessary. It is expected that run-of-river hydropower schemes are to be installed in water bodies which are most appropriate for them and are in places where impacts can be avoided or mitigated. Liermann & Roni (2008) stated that although our understanding of how salmonids interact with their freshwater habitats has steadily improved we are still a long way from being able to reliably predict population level effects of individual projects. This can only be improved however with effective monitoring and the cooperation of the developer, the designer and the various regulators. It is critical that researchers cooperate with managers to facilitate decision making and choose appropriate measures (Jonsson & Jonsson, 2011).

Although there are numerous potential measures developed to mitigate the impacts of run-of-river hydropower schemes (Chapter 2), there remains a poor understanding of the success of these measures, and when and where they are most effective (Gibbins *et al.,* 2008; Sabaton *et al.,* 2008). Minns (1997) stated that although attention is being paid to mitigate the potential negative effects of proposed hydropower developments, there is little attention or concern post development. Consequently, improvements in our understanding about the impacts and the efficiency of mitigation measures are slow. Unfortunately, due to the long and complex life cycle of many fish, such as the Atlantic salmon, only long-term investigations can reliably detect any substantial effects brought about by hydropower schemes upon fish populations (Ugedal *et al.,* 2006). There is thus a need to ensure that all impacts of existing and new hydropower schemes and measures to reduce the impacts are monitored using appropriate sampling strategies such as BACI analysis. This will indicate the status of fauna / flora in the proposed area, which can aid the choice of design and hydropower scheme best suited for the area. It will furthermore allow before and after comparisons, which will provide a clearer outline of the effects that are in response to hydropower operation and highlight the areas of concern.

Ultimately monitoring provides the best information to aid the choice of mitigation measure tailored to individual sites (Benstead, 1999), and can provide the information needed to adjust current management actions. It can also facilitate any needed project design or operational changes and benefit future EIA activities by providing for better monitoring plans. It is therefore vital that environmental monitoring programmes capable of detecting environmental effects are supported.

CHAPTER SEVEN

7. GENERAL DISCUSSION

7.1 Introduction

The flow regime of a river is crucial in supporting native biodiversity with natural changes between seasonal flows coinciding with natural life cycles; flow therefore provides the basis for a healthy and diverse aquatic ecosystem and timing of flows is ecologically critical (Poff *et al.,* 1997). Regardless of their size, hydropower schemes are generally associated with alterations and reductions in stream flow, which can potentially result in a substantial ecological impact (Anderson *et al*., 2006); the effects of flow alteration and loss of longitudinal connectivity are widely documented (Chapter 2). Natural variability in recruitment is a prominent feature of salmonids in UK rivers (Chapters 4 & 5) and therefore caution must be taken when attempting to assess the potential impact of run-of-river hydropower schemes on fish populations. With respect to each individual scheme, generally, trends that were observed at sites located within the abstracted reach mirrored those at sites located outside the abstracted reach (Chapters 4 & 5). Certain schemes however, such as Inverhaggernie Burn, indicated a potential impact of the run-of-river hydropower scheme on fish populations; densities within the abstracted reach declined over the study period post commissioning but the trend was not observed at the site located downstream of the abstracted reach. Execution of a BACI analysis (Chapter 6) improved understanding of monitoring requirements to produce statistically robust impact assessments and highlighted the necessity for long-term data collection. The overall aim of this study was to assess potential implications of ten run-of-river hydropower schemes (Chapter 3) on fish populations in Scottish streams, and ultimately develop a framework for future monitoring programmes. The study was divided into several topics each representing a different stage of the investigation; these were addressed in Chapters 2-6. This chapter integrates and discusses the knowledge gained from the previous chapters; key conclusions are drawn and recommendations for further studies are provided.

7.2 Potential impacts of run-of-river hydropower schemes on fish populations

The basic design of run-of-river hydropower schemes and the key impacts on fisheries were reviewed; possible mitigation measures were evaluated to address associated impacts (Chapter 2). A major problem associated with reviewing existing small-scale run-of-river hydropower schemes was the relative paucity of information on such schemes and the lack of empirical studies providing robust, defensible information of the various impacts identified. Nevertheless, many of the issues and concerns surrounding small-scale hydropower schemes were similar to those of the well documented impacts of large impoundment hydropower schemes; the principles of flow alteration and longitudinal connectivity were similar when assessing larger schemes and small-scale run-of-river schemes. Sabaton *et al.* (2008) reported that despite numerous habitat simulations being undertaken worldwide, there is currently still not enough biological monitoring being performed with regards to flow manipulation. While mitigation measures are incorporated into the design and operational requirements of run-of-river hydropower schemes, there remains a poor understanding of the success of these measures, and when and where they are most effective (Gibbins *et al.,* 2008; Sabaton *et al.,* 2008), consequently there is still a great deal of understanding to be gained. Minns (1997) stated that although attention is being paid to mitigate the potential negative effects of proposed hydropower developments, there is little attention or concern post-development. Consequently, improvements in our understanding about the impacts and the efficacy of mitigation measures are slow. Unfortunately, due to the long and complex life cycle of many fish, such as the Atlantic salmon, only long-term investigations can reliably detect any substantial effects brought about by hydropower schemes upon fish populations (Ugedal *et al.,* 2006).

7.2.1 Isolating natural variability from potential impacts of run-of-river hydropower schemes on salmonid populations

The implications of flow modification on fish are well documented (Chapter 2), yet it has been difficult in this investigation to relate changes in salmonid populations as a direct response to the commissioning of run-of-river hydropower schemes. The outcomes of the analyses of the fisheries surveys for the ten schemes under study were variable and dependent on design characteristics of the schemes and the sampling programmes carried out (Chapters 4 & 5). Furthermore, all schemes within the study had been operational for relatively short periods, between 1 and 3 years with the exception of Camserney Burn, which had been operational for 7 years; therefore analysis was restricted to the possible detection of potential impacts in the early stages of the schemes. It is worth noting that salmonid populations in streams may take up to 7 years (or two generations) to respond fully to environmental/habitat changes (Hunt, 1976); although the exploitation of Atlantic salmon has declined since the mid 1970s (Jonsson & Jonsson, 2011) there is still little recognisable positive effects on the stock abundances (Klemetsen *et al*., 2003) confirming that stocks can take decades to recover from a negative impact. Although Camserney Burn has been operational for 7 years, pre-commissioning data were unavailable; consequently fisheries data could not be compared pre- and post-hydropower commissioning, which caused limitations during interpretation of this data set.

From the ten schemes surveyed in this investigation it is clear that salmonid populations exhibit great spatial and temporal variation. In particular, salmonid populations revealed high variability in baseline data collected at a number of sites prior to hydropower schemes being constructed. For example, at site e in Keltney Burn, 0+ salmon density in 2005 was >160 fish/100 m^2, but between 2006 and 2009 (before hydropower commissioning) 0+ salmon density was generally stable ranging between 40-60 fish/100 m^2. Thus, 2005 appeared to be an exceptional year for 0+ salmon densities, but if this one year was used as the baseline against which to measure impact of a hydropower scheme, then subsequent monitoring surveys may suggest a high impact of flow regulation. This reinforces the importance of robust monitoring programmes, which are designed with the intention to account for spatial and temporal variation.

Section 7.2.1.1 provides a brief over-view of five of the ten run-of-river hydropower schemes and their respective key findings. These five schemes are considered to have had no impact on fish populations; observed fluctuation in fish densities were considered in response to natural variation. Section 7.2.1.2 provides a brief over-view of the remaining five schemes, each of these schemes have varying concerns related to the changes in fish populations observed and suggest possible impacts of run-of-river hydropower schemes albeit at one or two study sites. In some instances, populations have decreased at sites within the impacted reach but not at sites outside of the depleted reach. In other cases, monitoring undertaken for the scheme did not provide sufficient data which made it difficult to detect any impact and thus prevented definitive conclusions to be drawn.

7.2.1.1 Key findings from five run-of-river hydropower schemes displaying natural variation in fish populations

- Kinnaird Burn

  Although trout populations in the abstracted reach decreased in density during the years post hydropower commissioning, often to levels lower than those observed in years prior to hydropower commissioning, the trend in densities was mirrored at sites located outside of the abstracted reach. Therefore, the inter annual variations in salmonid densities made it difficult to detect any impacts, specifically in response to commissioning of the hydropower scheme, when comparing before/after and control/impact data.

- Keltney Burn

Both salmon and trout populations varied over the study period with similar trends in densities observed at sites located within and outside of the abstracted reach. Although in some years, densities of salmonids were higher in the abstracted reach (sites d and e), post commissioning, compared to years pre commissioning, an increase in densities was also observed at control site f, located downstream of the abstracted reach. At site d, in the abstracted reach, 0+ salmon and ≥1+ trout densities were higher in 2011 post commissioning than those observed in previous years pre hydropower commissioning. Similarly at site e, in the abstracted reach, ≥1+ trout densities were the highest in 2010 post commissioning than all years but also higher in 2011 compared to densities observed in 2008 and 2009 pre hydropower commissioning. As there is some protection of higher flows (in the winter months) it was considered that water abstraction had weakened spate flows and consequently reduced wash out of eggs, although it should be noted that data relating to the natural flow and abstraction rates are unavailable to confirm this. Nevertheless, the consideration that reduced flows had 'improved' the habitat and thus supported increased densities is perhaps unlikely given an increase was also observed at the downstream control site; suggesting other factors were responsible for the noted increase. It is possible that 2010 and 2011 were better years in terms of recruitment and thus have naturally provided higher numbers of 0+ individuals. Additionally, as no significant difference was observed in the wetted widths before and after commissioning of the scheme, it is suggested that the abstraction of water was not sufficiently severe to have a negative impact. Equally, it is possible that fish from other tributaries have moved into the reach of Keltney Burn and added to the populations in 2010 and 2011 resulting in higher ≥1+ densities.

- Innerhadden Burn

Densities of salmonids showed similar trends at sites located within and outside of the abstracted reach. ≥1+ trout populations at site d, located in the abstracted reach, were low during 2010 and 2011 post commissioning in comparison to earlier years pre commissioning. A similar trend however was observed at all control sites (a, c and f) located outside the abstracted reach. Densities of ≥1+ salmon were found to be higher in the abstracted reach at site e in 2011 post hydropower commission compared to some years prior hydropower commissioning, though this was also noted at the downstream control site f. Ultimately, trends in densities observed in the abstracted reach, mirrored those at the control sites. Consequently, the inter-annual variations observed in salmonid densities made it difficult to detect any impacts specifically associated with the Innerhadden Burn hydropower scheme.

- Camserney Burn

Although baseline data pre hydropower commissioning were unavailable, densities of salmonids were compared between sites within and outside of the abstracted reach. Salmon densities were generally low over the entire study period with the exception at site d outside the abstracted reach; the upstream reaches are not accessible and habitat in the lower reaches is considered unsuitable for salmon spawning or recruitment. Trends in trout densities observed at sites within the abstracted reach were mirrored at sites located outside the abstracted reach. At site c within the abstracted reach, densities of ≥1+ trout were higher than those at site d outside the abstracted reach over the study period with the exception of 2005 and 2009. While densities might have been higher at site c than d because the habitat is better, HABSCORE data were not collected at Camserney Burn and so this is merely a suggestion. Given similar trends in densities were observed in the abstracted reach and control sites, no impacts specifically in response to the Camserney Burn hydropower scheme could be detected. This interpretation should be treated with an element of caution however, as there are no pre commissioning data to definitively conclude this.

- Allt Gleann Da-Eig

Allt Gleann Da-Eig run-of-river hydropower scheme was commissioned post 2011 fisheries surveys; data were therefore analysed to assess any potential impact of the hydropower scheme's construction. 0+ trout densities at sites b, c and d (all downstream of the intake) declined in 2011 (post construction). Adult return rates however highlighted fewer adults returning in 2010 possibly explaining the reduced number of 0+ trout in the subsequent year. It should be noted however that residential brown trout may also have contributed to the recruitment of 0+ fish and so the relationship between the decreasing numbers of 0+ fish to the reduction of returning sea trout is indicative only. It should be noted, however, that densities at site e, downstream of the outfall (thus also subject to potential impacts of construction) were higher in 2011 (post construction) than 2007 (pre construction). This suggests that the declines observed at sites b, c and d are perhaps more likely to be in response to natural variation. In some instances densities were higher at sites located downstream of the intake to that above the intake. That said, while ≥1+ trout populations were higher at site b than densities observed at site a, this was true for all years of the study period both pre and post construction, with the exception of 2011. It is considered that this could just be due to better habitat at site b. Ultimately, fluctuations observed in salmonid densities at sites downstream of the intake could not be attributed specifically to the construction of Allt Gleann Da-Eig hydropower scheme.

7.2.1.2 Possible impacts of run-of-river hydropower schemes on the fish populations

- In Rottal Burn at site e, in the abstracted reach, ≥1+ salmon populations appeared to have decreased in 2010 and 2011 post hydropower commissioning compared with earlier years pre hydropower commissioning. This declining trend in ≥1+ salmon densities was not mirrored in densities located at control site g, downstream of the depleted reach or at site f (also located in the depleted reach but further downstream). Thus, reduced flow in the depleted stretch of river due to the Rottal Burn run-of-river hydropower scheme *could* be responsible for the decline in ≥1+ salmon densities. It is possible that a reduction in water depth and/or velocity has altered the habitat, such as the substratum composition (which is an important factor for salmon parr rearing), and thus reduced the habitat suitability for ≥1+ salmon. Unfortunately HABSCORE data were not collected at site e and so this cannot be definitively confirmed. While the reduction of ≥1+ salmon densities could equally be due to natural fluctuations it should be highlighted that this reduction was not observed at site f (depleted reach) or g (downstream control). Although site f is also located in the depleted reach, this site is further downstream and so the characteristics of this site are different in comparison to site e. Ultimately, it is possible that a combination of factors, including the alteration of habitat and low flows has potentially affected ≥1+ salmon densities (0+ fish densities did not decline at this site).
- In Inverhaggernie Burn at site c, located in the abstracted reach, 0+ salmon populations appeared to have decreased in 2011 compared with earlier years pre commissioning; the trend was not observed at the site located downstream of the depleted reach. However, it is not clear if this decline was a natural phenomenon (though the decline was not observed at the control site), a result of flow regulation (densities only declined in the abstracted reach), or the presence of a weir at the outfall. The results from this scheme have indicated the importance of maintaining longitudinal connectivity; both physical (weir) and non physical (reduced flow) barriers can contribute to impairing a rivers' connectivity and thus inhibit salmonids movements.
- In the upper reaches (located in the abstracted reach) of the River Callop there was a decrease in 0+ and ≥1+ trout densities over the study period. It could not be identified if this was a natural population decline or due to flow regulation as

there were no upstream control sites monitored prior to the scheme commencing abstraction. While the decline in trout populations was observed at site c1 downstream of the abstracted reach possibly suggesting a natural population decline, the upstream and downstream reaches of a river can be different in characteristics and thus the fish populations that they support. Therefore it is perhaps not ideal or reliable to use control sites located in the downstream reach as a control for sites in the upper reaches and as such has highlighted the necessity for independent control sites located both in the upper and lower reaches of a river. Ultimately, there were not sufficient data available to definitively conclude whether the decline observed was in response to the hydropower scheme or natural variability.

- In Ardvorlich Burn it was difficult to detect any impact of the hydropower scheme on salmon, trout and eel populations due to the limited data available. As the scheme commenced abstraction in 2011 any impact may not initially be detected from surveys carried out in the same year. As mentioned, salmonid populations may not respond to environmental changes for up to 7 years (Hunt, 1976) and so it is extremely unlikely that any impact would be recognisable within a 12 month period. Furthermore, as there is such great natural variability in fish populations, even if a negative response were to be found it would be difficult to associate it directly to the commissioning of the hydropower scheme based on 1 year's post-commissioning data. Consequently, the monitoring design implemented for the Ardvorlich Burn hydropower scheme has made it impossible to draw a definitive conclusion and has specifically highlighted the importance of post monitoring data.
- Trout population densities declined at site D6 in Douglas Water from the first survey in 2002 to the second survey in 2011, but this decline was also found at site D8, the control site upstream of the abstracted reach. At the other sites within the abstracted reach trout densities were higher in 2011 than 2002, but the time elapsed between the baseline surveys in 2002, commissioning of the scheme in 2008 and subsequent monitoring surveys 3 years later in 2011 makes identifying any impact due to flow regulations tremendously difficult. As discussed in Chapter 4, salmonid populations display great inter-annual variability and therefore the lack of data has made it difficult to draw a definitive conclusion. The monitoring implemented for the Douglas Water hydropower scheme has reinforced the importance of consecutive annual pre and post monitoring surveys.

7.2.2 Disruption to longitudinal connectivity

Small hydropower schemes are often claimed to have minimal impacts on the environment and are even portrayed as favourable to ecosystems and fish. For example, the British Hydropower Association and the European Small Hydro Association websites assert that “small-scale HP is environmentally friendly”, it “respects and protects the environment” and “with run-of-river schemes little reservoir impoundment takes place, and the little dams create little pounds which are very favourable for ecosystems, fish and water storage.” Fraenkel *et al.* (1991) concluded that “micro, or small-scale hydro, is one of the most environmentally benign energy conversion options available because, unlike large-scale hydro, it does not attempt to interfere significantly with river flows*”.* These claims are not, however, supported by empirical evidence, and there is the possibility that fish populations are negatively affected, particularly salmonid fish populations in relation to depleted reaches (Chapter 2) and migration.

Longitudinal riverine connectivity has been recognised as crucial to the functioning of river ecosystems on a basin scale (Ward, 1989). Any impoundment, whether just a few centimetres or a hundred metres high, may thwart or delay fish passage and restrict access to certain areas of river (Peter, 1998; Paish, 2002). One of the most prominent results from this study was the decline in 0+ salmon densities at Inverhaggernie Burn site c (within depleted reach); a weir that was constructed at the outfall (at a later date post commissioning) could perhaps be responsible for this. Impounding structures can block or delay the movements of migratory fish, and are responsible for the decline or extirpation of many native salmon populations in both the Atlantic (Netboy, 1968; Mills, 1989) and Pacific (Meehan, 1991; Frisell, 1993; Levin & Tolimieri, 2001). It is commonly documented that disruption to longitudinal connectivity can preclude migrations of many fish (Calles & Greenberg, 2005), restricting access to spawning grounds; indeed Gowans *et al.* (2003) reported a cumulative effect of multiple barriers on upstream migration of salmonids. It is therefore a possibility that the presence of the weir at Inverhaggernie Burn could have obstructed adult salmon from accessing upstream spawning habitat at site c (above the weir), reflected by the reduced recruitment of 0+ salmon in 2011. This is supported by Gosset *et al.* (2006) who demonstrated that the presence of a weir prevented brown trout from accessing their spawning destination. Brown trout appeared to select only one tributary for spawning, yet when confronted with an obstacle such as a weir, they were reluctant to enter alternative tributaries, thus confirming selective behaviour in relation to spawning destination (Gosset *et al.*, 2006). Webb (1990) also found that salmon that failed to pass an obstacle did not return to the structure. Equally, the weir at Inverhaggernie Burn could have restricted the number of 0+ salmon, that were spawned downstream of the outfall (site d), moving upstream to utilise the habitat at site c. Indeed, even small technical constructions with a head loss of only 25 cm inhibit the movement of young trout and other small fish species (Peter, 1998).

In spite of this, the reduction in densities could equally have been in response to a natural phenomenon; however, it is important to note that populations at site d downstream of the abstracted reach did not decline in 2011. For that reason, it is considered unlikely that the reduction in densities observed in the depleted reach of Inverhaggernie Burn was due to natural fluctuations and rather in response to either reduced flow in the abstracted reach or presence of the weir; both factors result from the commissioning of Inverhaggernie Burn run-of-river hydropower scheme.

Chapter 4 highlighted some schemes where reduced numbers of fish were observed above the intake. While a site located above the intake acts as a control against flow regulated impacts, there is the possibility that fish located at sites upstream of intakes are affected by disruption to longitudinal connectivity. The main issue is that fish have to negotiate the weir in order to utilise the habitat upstream, however, this task is exacerbated further by reduced water downstream of the weir as a result of the hydropower schemes abstraction. Lucas & Baras (2001) stressed that even small impounding structures, such as the weirs constructed for the schemes within this study, are equally problematic to maintaining open migratory pathways as larger dams if hydraulic conditions associated with the barrier are inappropriate. Often precise conditions are required at each barrier to enable successful passage, and modification of flow regimes by hydropower schemes may increase the severity of obstructions in terms of upstream migration (Solomon, 1992; Sambrook & Cowx, 2000; Anderson *et al*., 2006; Kemp *et al*., 2010). The water depth downstream (and upstream) of potential barriers also influences passage success. For example, a 45-cm high vertical sill is insurmountable for salmonids if the water depth downstream is not sufficient (Ovidio & Philippart, 2002). Water depth downstream of the intake will be reduced as it is located in the depleted stretch of river, thus it is possible that there has been insufficient water (during the spawning migration period) to allow fish to negotiate the weirs leading to reduced densities at upstream control sites.

While weirs can elevate water levels upstream which may benefit some animals, including fish, hydraulic conditions are greatly altered by the construction of impoundments (Fjellheim & Raddum, 1996). For example, weirs can prevent gravel renewal leading to a reduction in the quality (e.g. increased content of fines) and area of gravel shoals downstream (Kondolf, 2000), which are vital for spawning by salmonids. This factor could also contribute to the explanation of why there were reduced numbers of 0+ salmonids at sites located downstream of the intake at both the River Callop and Inverhaggernie Burn.

In order to ease the negotiation of weirs, many schemes are constructed with fish passes. Currently, however, passage rates and thus the efficiency of these passes are unknown. In order to successfully attract fish to the entrance and indeed allow safe passage, fish passes must be designed in a specific manner. The Environment Agency “Fish pass manual” provides information on the statutory processes of fish pass provision and also details the types of passes available (Armstrong, *et al.*, 2010). Although SEPA’s “Guidance for developers of run-of-river hydropower schemes” also includes information regarding fish passes there is no protocol for checking each fish pass has conformed to the necessary requirements. SEPA do not sign off the overall fish pass design, nor are they responsible for following up each fish pass post construction; an approach undertaken by the EA. A fish pass constructed at the Allt Gleann Da-Eig intake (Figure 7.1) illustrates some of the issues, such as the positioning of the entrance and lack of attraction flow, associated with fish pass designs. Where necessary, for the purpose of this study, the design is assessed according to the EA manual opposed to SEPA; the latter offers limited description of exact design requirements.

It is of great importance that fish can ascend and/or descend fish passes safely, critically at specific times of the year when migration movements are essential and fish passes are needed to allow migratory fish to swim upstream and reach their spawning grounds (Jungwirth, 1996; Knaepkens *et al.,* 2005). Delaying downstream or upstream migration, during periods of time when fish are trying to negotiate or find a fish pass, can potentially cause knock on effects such as increased predation and diseases (Mathers *et al.,* 2002); this could ultimately reduce the status of fish populations. An important feature of a fish pass is the location of the entrance and attraction flow (Clay, 1995). It is considered the Allt Gleann Da-Eig fish pass (Figure 7.1) meets the EA criteria in terms of location - passes situated near a bank side are preferred to facilitate monitoring and maintenance, and are to be located at the most upstream point of the barrier (Armstrong *et al.*, 2010). However, the exact location of the fish pass in relation to the predominant flow of river must also be taken into account to facilitate fish finding the entrance. Indeed, the fish pass could be deemed ineffective unless fish can find and negotiate it. When travelling upstream, individuals will follow the predominant flow of the river, thus the fish pass must have a relatively large attraction flow to help fish locate its entrance (Cowx & Welcomme, 1998; Trussart *et al.,* 2002). The majority of flow here however is down the main river channel and to the left hand side (Figure 7.1a & b); thus the opposite end to the fish pass entrance, located at the far right hand side of the weir.

Figure 7.1 Fish pass constructed at Allt Gleann Da-Eig hydropower scheme. Photographs were taken September 2011.

Considering the fish pass is targeted towards adults it is very narrow which reduces chances further of fish finding it, furthermore picture d shows a boulder located just beneath the entrance which not only hinders the attraction flow but would possibly make it difficult for fish to ascend. According to SEPA guidelines the depth of the plunge pool on the downstream side of the intake must be 1 m or 1/3 the height of the weir (whichever the smallest) in order for fish to leap over the crest of the weir (SEPA, 2010a); it is unsure whether the depth of this plunge pool conforms to these requirements.

7.2.3 Conclusions

In all cases, there was considerable inter-annual variation in recruitment of trout, and where present Atlantic salmon, typical of that found in fish populations (Crisp, 1993). In the majority of cases, where the sampling regime allowed, similar trends in densities within and outside the impacted reach were found. In all cases it was difficult to ascertain if the hydropower scheme was the direct cause of the changes observed, whether there were indirect effects, e.g. as a result of reduced flows in the depleted reach prohibiting migration movements, or whether they were largely due to natural fluctuations in recruitment.

The importance of maintaining longitudinal connectivity for migratory adult salmon and trout has been raised throughout this thesis. It is considered a priority given the potential impacts of the weir at Inverhaggernie Burn and also the possibility that weirs constructed at the intake could have potentially hindered upstream migration of adult fish. Two main problems seem to arise. Firstly, it is considered that a reduction of flow in the depleted reach prevents adult fish accessing spawning habitat in the upstream reaches. It should be noted however, that this is pure speculation as there are no flow data available, and ideally, adult fish count data would be used in hand with flow data to confirm this. Secondly, it is not known whether fish are able to negotiate fish passes to access the habitat upstream of the intake weirs and whether this problem is being exacerbated by reduced flow downstream of the weir in the depleted reach. Both of these suggestions are supported by literature. It is documented that effects from hydrological alterations are evident on fish (Nilsson & Brittain, 1996; Welcomme *et al.*, 2006), where changes in aquatic ecosystems can restrict or hinder fish migration, affect water quantity and quality, increase predation and cause direct damage and stress (Schilt, 2007). While some schemes however are located in steep sections of rivers or above impassable waterfalls and cascades, there are also many that are constructed in reaches associated with spawning and nursery areas within or upstream of the depleted reach. Indeed, there are good numbers of salmon located in the depleted reach of the Keltney Burn and Rottal Burn and similarly for trout located in the depleted reach of Camserney Burn. While it could therefore be suggested that reduced flows in the depleted reach eased the severity of current velocity and reduced the number of eggs washed out, it is also a possibility that perhaps abstraction was not sufficiently severe to have a negative impact, hence the lack of any detectable impact. However, as flow data were unavailable, regarding both abstraction rates and remaining flows above/below and within the depleted reach, it is incredibly difficult to assess these relationships and draw definitive conclusions.

7.2.4 Recommendations

Although several attempts have been made to identify the reasons for possible declines in fish populations, there are limitations within this study. Due to hydrological data being unavailable, relationships between fish populations and flow regimes could not be analysed and therefore definitive conclusions could not be drawn in most cases.

Although flow data are available representing the volume of water abstracted, there are no flow data relating to the amount of water that is actually left within the depleted stretch of river during periods of abstraction. **It is recommended that on future hydropower schemes gauges are installed on the intake and above and below the outfall to provide information about the proportion of flow that is abstracted in relation to the natural flow**. This may allow direct comparisons to be made between flow and ecological data providing a more comprehensive analysis and greater understanding of population responses to flow alterations, especially within the depleted stretch of river.

It has been suggested in this study that possible reductions in flow have eased the severity of currents in the depleted reach and in turn *may* have reduced the number of eggs that are washed away resulting in an increase in 0+ individuals in the depleted reach. **It is suggested that detailed habitat assessments are undertaken in future studies to compare the habitat between sites located within and outside the depleted reach and later assessed in relation to flow data**. Furthermore, although this study has indicated some reaches to be good spawning and nursery areas, with consistently good numbers of 0+ fish, it is unclear whether these numbers are in proportion to the number of spawning redds initially built across the reach. Indeed, redds located outside a depleted reach could be washed out in contrast to those in the abstracted reach. By contrast they may also be left exposed by lower flows at critical periods. **Spawning redds should therefore be identified and monitored.** Spawning redds in reaches associated with flow alterations such as Keltney Burn should also be compared to redds located in reaches with a natural flow regime to allow comparisons between hatching time, temperature, O_2 concentrations, sediment and pH level. Walk over surveys should be conducted before the start of the spawning period with details noted on numbers and location. Tagging fish would also allow greater knowledge to be gained; intricate details such as timing of arrival and departure at spawning grounds, selection of gravels and habitat preference and flows associated with redd building could be gained. This information could be used to assist guidance of flow requirements and duration/timing of shut down periods.

The findings of this study indicated concerns regarding longitudinal connectivity. Barriers were considered to be a possible contributor to the reduced numbers of 0+ individuals in the depleted reach of Inverhaggernie Burn. **It is recommended that natural variation between seasonal flow regimes and longitudinal connectivity are protected wherever possible to support migration movements of salmonids**. While it is important to consider low flows it is equally important **that research is targeted into the abstraction of much higher flows, such as spate flows that are**

fundamentally important for migrating species to overcome barriers and access spawning grounds. Hydrological data provided by gauges may allow flow duration curves to identify spate flows; these should then be analysed in relation to the movements of tagged fish through reaches associated with hydropower schemes.

It is commonly documented that the incubation period of salmonid eggs can be heavily influenced by water temperature (Berg & Moen, 1999; Jonsson & Jonsson, 2011); the duration of the alevin period increases with decreasing water temperate and as such colder water leads to a longer incubation period. As there is a reduction of water in the depleted reach it is unknown whether any increase in temperatures are causing significant changes to spawning efficiency in the depleted reach of river. **It is recommended that temperature profiles are monitored in relation to the schemes to assess the spawning process, such as egg incubation and hatching time, in relation to temperature and flow**.

Changes in flows related to the operation of hydropower schemes must ensure important habitats such as nursery and spawning areas are protected, especially during critical periods of salmonid life cycles. It is crucial that certain flows during the year are protected to ensure successful recruitment and migrations of salmonids. As flow requirements are essential to the development and survival of embryos **future studies should attempt to investigate the impact of prolonged reduced flows, as experienced in the depleted reach of a river and the subsequent effects during egg incubation**. This could be achieved under fluvarium conditions exposing eggs to varying flows; eggs should be monitored until fry emerge and any effects recorded. Although many of the schemes in this study ceased abstraction at Q_{85} this is not true of all schemes that are being granted licences and the HOF can vary.

Future incubation studies should aim to investigate the effects of sedimentation during the spawning and incubation period and whether this process is elevated within the depleted stretch of a river due to reduced flow. This could be assessed by simply comparing levels of any sediment surrounding redds located in a depleted stretch of river to those in a reach with no water abstraction. It is recommended flow data are collected simultaneously to assess the optimal flow needed to clear spawning redds of any sediment for effective recruitment. A flow meter should be used to record the flows at varying depths of the water column and in the gravel.

Although fish passes can vary in their design this study has highlighted some concerning factors that are associated with current fish pass designs and there is currently relatively little literature available documenting the efficiency of such fish passes. While fish passes are usually not a high consideration in the design of high-head run-of-river hydropower schemes they are useful on some large intake weirs and certainly for low head run-of-river hydropower schemes. **It is suggested future research aims to identify if there are any population-level impacts from the disruption to upstream and downstream migrations of fish. Monitoring programmes could establish if fish are able to move up and downstream freely and most importantly safely. It is further recommended that investigations into the behaviour of different fish species at different fish pass types is targeted, including both salmonids and cyprinids, as the latter can sometimes be overlooked.** Furthermore, it would be ideal to discover where fish passes are currently most needed i.e. on rivers where hydropower schemes are proposed with good quality fish stocks. If longitudinal connectivity is impaired, a loss of recruitment to the system can be seen; **therefore fish passes should be installed as a requirement of the scheme**. It is a necessity for fish passes to be designed correctly and approved, on this basis it is recommended that **mitigation measures must be implemented properly and subsequently checked by SEPA or EA post construction.**

7.3 Monitoring programmes of fish populations

Ten run-of-river hydropower schemes were monitored in this study, with slight differences between some schemes regarding the sampling programme. In the majority of instances, monitoring of schemes provided sufficient data sets incorporating spatial and temporal replication; these were considered to be Kinnaird Burn, Keltney Burn, Innerhadden Burn, Inverhaggernie Burn, Rottal Burn, Camserney Burn and Allt Gleann Da-Eig. Although there were limitations regarding data analysis with Camserney Burn and Allt Gleann Da-Eig, restrictions were not caused by insufficient sampling programmes. Pre commissioning data were unfortunately unavailable for Camserney Burn and problems with Allt Gleann Da-Eig meant commissioning of the scheme was delayed past the proposed date; consequently the scheme became operational post the 2011 fisheries surveys. While both of these factors were outside the control of this study they highlight the reality of potential drawbacks that can arise.

The outputs however, raised a series of issues regarding existing monitoring programmes on some of the schemes that need to be addressed if robust impact assessments are to be undertaken in the future.

- Considerable differences were found in the EIA monitoring strategies adopted for the different schemes under study. While many of the schemes detailed in Chapter 4 followed a very similar format (in terms of the number and location of control and impact sites) for the author’s research, there were great differences by comparison in the sampling programmes for other schemes such as the River

Callop, Douglas Water and Ardvorlich Burn undertaken by consultancies and fisheries trusts (Chapter 5); again in terms of the number and location of control and impact sites. Some schemes for example, had double or triple intakes (Kinnaird Burn, Rottal Burn, Innerhadden Burn) and while the sampling regime at these schemes incorporated one control and one impact site located at both of the intakes, the sampling at Douglas Water had one control and one impact to represent both intakes; one control site at the first intake and one impact site at the second intake. Consequently, data interpretation at Douglas Water was limited. While it was therefore not possible to make robust comparison between schemes to determine whether generic impacts occur, or the sources of any impact, the attempt to try and assess generic impacts could perhaps be irrelevant given that the study has portrayed great variability in fish populations both annually at each scheme and also between rivers.

- Often in schemes undertaken by consultancies and fisheries trusts no pre-scheme baseline data were collected e.g. Camserney Burn, Douglas Water and Ardvorlich Burn or the surveys were outdated (undertaken too many years prior to the scheme becoming operational). For example, the only pre hydropower commissioning fisheries surveys for Ardvorlich Burn were conducted in 2008 yet the scheme did not become operational until 3 years after in 2011. Similarly, the first and only pre hydropower commissioning fisheries surveys for Douglas Water were conducted in 2002 but the scheme was not commissioned until 6 years later in 2008. Therefore they are considered invalid for accurate assessment of impact, especially given the large natural fluctuations in fish population densities encountered.
- In some cases such as the River Callop and Douglas Water, fish surveys were only undertaken in the impacted, depleted, reach with no surveys above or below the intake or outfall to act as control, thus data interpretation was limited.
- In some cases there was insufficient post-scheme monitoring to discriminate natural population fluctuations from the impact of the scheme. Many schemes were commissioned in 2009/2010 thus providing only a couple of years post-commissioning data to identify any impacts before completion of this study. Ardvorlich Burn was commissioned in 2011 shortly before the 2011 fisheries surveys; thus only one post-commissioning data set was available for interpretation of the impact of a scheme that had been operational for less than a year.
- There was little diversity in the type of schemes under study, thus limiting interpretation of the scale of impact. For example, all the schemes within this study are generally small scale in terms of their energy output and many ceased

abstraction at Q_{85}. It must be noted however, that these schemes are representative of the types of schemes that have been licenced in Scotland over the past decade.

- All of the run-of-river schemes monitored in this study were high head and thus interpretation of the impacts are not transferable to low-head schemes that are often located in lowland rivers.

As salmonid population densities in streams can vary considerably from year to year this should be considered when interpreting the results from a "one-off' electric fishing survey, such as those undertaken for the Ardvorlich Burn and Douglas Water hydropower schemes. If the objective is to set a pre-intervention baseline for comparison with the situation after intervention then it is important for both pre- and post-intervention studies to cover an adequate number of years.

A further concern that arose from interpretation of the results was use of the different fish classification schemes for detecting large-scale change in abundance. Considerable differences were found in classification of the status of the salmon and trout populations using the different national schemes and efforts need to be made to identify causes for the deviations observed and adopt the most suitable methodology as the standard tool. Out of the five classification schemes used – SFCC national, SFCC regional, SFCC national based on river width, SFCC regional based on river width and EA-FCS – it is suggested that the SFCC regional based on river width classification be used in the future. This is considered to be the most appropriate purely because it is less likely to over/under-estimate populations as it is the most refined in terms of location (regional rather than national) and also specific to width, when classifying fish populations. At site d (abstracted reach) in Kinnaird Burn, the SFCC national, SFCC regional, and SFCC national based on river width classified 0+ trout populations in 2008 as fair/average (class C) whereas the SFCC regional based on river width classified the population as poor (class E). Although the former three methods classify the populations in the same category (class C) it could be argued that they are over-estimating the populations as they are specific to neither region nor river width. The EA scheme classified 0+ trout populations in 2008 as good (class B) which again is not specific to a particular region or river width. Additionally, for the purpose of this thesis, as all hydropower schemes are located in Scotland, fish populations should ideally be classified using a Scottish classification scheme as opposed to an English method if only one is to be used.

It has become increasingly apparent through the investigation that some existing sampling protocols and impact assessments were inadequate to provide robust, defensible information about the impact of run-of-river hydropower schemes on fisheries. Schemes that had little fisheries data on which to construct an impact statement include the River Callop, Douglas Water and Ardvorlich Burn, and are detailed in Chapter 5. To address this potential problem, it was considered imperative for a robust monitoring protocol to be established for the future so that potential impacts of future run-of-river hydropower schemes could be assessed as fully as possible, without any limitations during data analysis stages.

7.3.1 BACI analysis

Chapter 6 illustrated how a BACI analysis can be used to produce a framework for future monitoring programmes to assess the potential impact of run-of-river hydropower schemes. Implementing an effective monitoring design with sufficient control and impact sites ultimately accounts for the cautions linked with impact assessments by isolating the environmental impact from natural variability, thereby allowing for statistically robust impact assessments to be conducted. The outputs confirm that long term monitoring is crucial if an impact, with a determined level of statistical significance, is to be detected. Based on resource calculations **it is recommended that surveys are performed 3 years before and 3 years after commissioning of a run-of-river hydropower scheme with four control and four impact sites to be surveyed** (Tables 6.3-6.5). A minimum of 2 years was required in order for the actual variance to be below the target variance, but, given the amount of temporal variation that fish populations display (evident in this study) a minimum of 3 years is suggested to account for natural temporal fluctuations. The number of control and impact sites that would be required, (based on 3 years' pre/post monitoring) varied for each of the three outputs (Tables 6.3-6.5). Consequently, the average of the three outputs was suggested, with four control and four impact sites to be surveyed (Chapter 6).

Although BACI analysis works well, in reality there are several financial and commercial restraints attached. Given the current concerns surrounding global warming there is great demand for renewable energy; run-of-river hydropower schemes being an important contributor. It may therefore be unrealistic for hydropower developers to wait 3 years before a scheme may be commissioned given the timescale in which renewable energy targets need to be achieved. Additionally, if 3 years of baseline data were advised, it would dramatically increase financial costs associated with the scheme, regardless of who the financial responsibility laid with, i.e. the hydropower developer or governing body. While the sampling regime derived from the BACI analysis should be considered when the necessary resources are available to do so, the monitoring programmes used in many of the schemes in this investigation

(Kinnaird Burn, Keltney Burn, Innerhadden Burn, Inverhaggernie Burn, Rottal Burn, Camserney Burn, Allt Gleann Da-Eig) were considered optimal (but sub-optimal according to BACI) given the resources and time available.

As a minimum however, when resources and time are limited, 2 years of baseline data must be carried out prior hydropower commissioning to account for temporal variation. Based on the resource calculations of 2 years pre and post monitoring, it is advised that five control and five impact sites are monitored to collect sufficient data to perform an impact assessment capable of detecting 50% change in fish density.

BACI analysis was further used to determine if there had been an impact on the fish populations at Keltney Burn due to the commissioning of the run-of-river hydropower scheme. Determined by the outputs of the resource calculation, a full impact assessment could only be conducted on the 0+ salmon populations. The mean change in 0+ salmon density at the sites located within the depleted lower reach of Keltney Burn after flow modification was -0.24 ± 2.44 fish/100 m^2. The confidence limits enclose zero therefore **the null hypothesis was accepted, i.e. there were no significant differences in the 0+ salmon populations in Keltney Burn before and after the changes in flow regime that can be attributed to the commissioning of the hydropower scheme.**

In the upland reaches of the rivers studied in this investigation Atlantic salmon were absent and the densities of trout were relatively low, thus it would have been very difficult to detect an impact and furthermore relate any change within populations to the commissioning of hydropower schemes; as such a BACI analysis was not performed using the densities of trout from upland reaches. Small populations such as those typically found in upper reaches are much more vulnerable to natural influences than larger populations and are more susceptible to natural variation due to fluctuating conditions, such as low food productivity and environmental factors. This however does not de-value their status or importance to the ecosystem. Although trout populations are generally sparse in the upstream reaches **it is recognised that these populations are isolated, and therefore may represent an important genetic strain unaffected by stocking; consequently they should be protected wherever possible.** The diverse life history of salmon and trout has been suggested as the mechanism that enables small populations to persist (Saunders & Schom, 1985). Furthermore upstream brown trout populations may add significant value to the brown trout populations found in downstream reaches.

7.3.2 Conclusion

The main conclusion from the field studies was that a proportion of the EIA surveys (River Callop, Ardvorlich Burn and Douglas Water) carried out were insufficient for the purpose of this study and potentially did not meet the Scottish government or SEPA's criteria for assessment. The reader is referred to the Water Environment (Controlled Activities) (Scotland) Regulations 2011. Essentially, although the monitoring design of each scheme accounted for some control and impact sites across the relevant reach of river some were insufficiently located, for example there was no control site in the River Callop representing the upper reaches and therefore interpretation of the data set was limited; consequently conclusions on the potential impact of the hydropower scheme on the fish populations should be treated with caution. The study highlights a need for better guidance on survey design criteria to be issued by regulatory authorities for robust data analysis and impact assessments to be conducted in the future.

7.3.3 Recommendations

This study has highlighted the variability between run-of-river hydropower schemes; each scheme is specific in its design and location and thus potential impacts have differed and varied in scale. The sampling protocols for each of the schemes under study also varied considerably and it is considered that some provided inadequate pre application and post commissioning monitoring of schemes. **It is recommended that both SEPA and the EA ensure that the likely proliferation of run-of-river hydropower schemes is accompanied by an effective monitoring programme to ensure that any unforeseen adverse effects are addressed and rectified**. **It is recommended that protocols for monitoring the status of fish populations include both pre-and post-operational surveys.** Monitoring should preferably be over several generations (6 years in total, 3 years pre and post hydropower commissioning) to account for inter-annual variations in fish populations.

Outputs from a BACI analysis indicated that current monitoring surveys are generally inadequate to perform impact assessments if a 50% change in fish density is to be detected. Resource calculations therefore identified how much monitoring spatially and temporally would be needed to allow impact assessments to be performed (3 years before, 3 years after with 4 control and 4 impact sites). **It is crucial that years sampled are continuous** to reduce the potential influences of temporal variation. Having continuous data sets will considerably strengthen the interpretation of potential impacts of a scheme and provide more statistically robust data. From the variability in current monitoring designs it is clear that control and impact sites are vital in the production of an effective monitoring programme to reduce variance and allow statistically robust assessment to be performed. **Annual fisheries surveys should be performed at the same sample sites at the same time each year to provide comparable data** and allow any unforeseeable long term impacts related to run-of-river hydropower schemes to be indicated. In addition monitoring should also ensure mitigation measures to minimise any impact, such as fish passage facilities and screening are functioning as expected and mortality caused by entrainment through the turbines or impingement on the screens is negligible.

It is recommended the resource calculation is performed after the initial 3 years (pre hydropower commissioning sampling) to confirm sufficient data have been collected to perform an impact assessment. This is based however on the assumption that spatial and temporal variation will remain the same during the 3 years of sampling post hydropower commissioning.

While the recommendations produced from the BACI analysis are ideal when there are no funding and time constraints, it may not be feasible for hydropower developers to wait for 3 years before their scheme is licenced, especially given the push for schemes by the government. As such, using outputs further produced from BACI, **a minimum of 2 years' baseline data, five control and five impact sites should be sampled to collect sufficient data and execute impact assessments**.

Although fisheries surveys are currently required under the CAR licence, details of exact requirements are limited. This study has provided detailed insight into the variability of current monitoring designs implemented on run-of-river hydropower schemes. Surveys undertaken by HIFI (with the exception of Keltney Burn), LFT, SEPA and MFC did not provide the data at a level that allowed a BACI analysis to be performed and indeed some prohibited aspects of data analysis; as such **it is recommended that more detailed survey designs are required and are subsequently made available for developers to allow them to provide appropriate information for robust EIAs.** These should, at minimum, investigate juvenile fish densities in the reaches above the intake, below the outfall and where appropriate in the depleted flow reach.

It is crucial that sufficient fisheries data are collected prior to regulatory authority consent, to provide information on potential impacts, and operation of the hydropower scheme to provide a baseline against which to measure any short or long-term impacts. This is essential to allow adjustments or amendments to the scheme design if deterioration of fish stocks is detected. When undertaking a monitoring programme to assess the impact of run-of-river type hydropower schemes on fisheries a number of questions need to be answered. These include:

- What is the population structure and density of the resident fish stocks and how will they be potentially affected?
- How important is the impacted reach as migratory and non-migratory salmonid spawning and nursery areas?
- What contribution does the impacted reach make to the overall recruitment of fish in the river catchment?
- Will the hydropower scheme:
    * impede migration of adult salmonids and downstream migration of smolts?
    * impede the migration of other freshwater fish species, especially those of high conservation value?
    * impede the dispersal of juvenile fish?
    * cause mortality of juveniles and smolts through entrainment, particularly during the dispersal stages of the life cycle?
    * impact on eel populations?
- What will be the effect of the operating rules on the flow regimes and how will this affect the fisheries with respect to:
    * potential loss of spawning and nursery habitats either on a temporary or permanent basis?
    * changes in wetted area and available food resources?
- What will be the losses of amenity value in terms of fisheries and other recreation and conservation aspects?

Detailed sampling strategies should indicate the status of fauna / flora in the proposed area, which can aid the choice of design and hydropower scheme best suited for the area. It will furthermore allow before and after comparisons, which will provide a clearer outline of the effects that are in response to hydropower operation and highlight the areas of concern. Ultimately it will provide the best information to aid the choice of mitigation measure tailored to individual sites (Benstead, 1999).

This study has confirmed the Tay catchment to provide high quality reaches of nursery and spawning habitat indicated by the annual recruitment of 0+ salmonids. **While already a feature of SEPA guidance, it is crucial that monitoring be of priority when schemes are located in sensitive areas (i.e. areas associated with salmonid migration and spawning and equally on rivers designated as SAC);** monitoring should target specific indicator species of environmental quality such as salmon and the impact of the scheme on fish migration through tagging and counter methodologies is advised.

REFERENCES

Aarestrup, K. & Koed, A. (2003) Survival of migrating sea trout (*Salmo trutta)* and Atlantic salmon (*Salmo salar)* smolts negotiating weirs in small Danish rivers. *Ecology of Freshwater Fish* **12,** *169-176.*

Aarts, B.G.W.,Van Den Brink., F.W.B. & Nienhuis, P.H. (2004) Habitat loss as the main cause of the slow recovery of fish faunas of regulated large rivers in Europe: the transversal floodplain gradient. *River Research & Applications* **20,** 3-21.

Abernethy, C. S., Amidan, B.G. & Čada, G.F. (2002) Simulated Passage Through a Modified Kaplan Turbine Pressure Regime: A Supplement to “Laboratory Studies of the Effects of Pressure and Dissolved Gas Supersaturation on Turbine-Passed Fish.” PNNL-13470-A, Pacific Northwest National Laboratory, Richland, WA.

Acornley, R.M. & Sear, D.A. (1999) Sediment transport and siltation of brown trout (*Salmo trutta* L.) spawning gravels in chalk streams. *Hydrological Processes* **13,** 447-458.

Acreman, M. & Dunbar, M.J. (2004) Defining environmental river flow requirements - a review. *Hydrology and Earth System Sciences* **8,** *861-876.*

Acreman, M.C. & Ferguson, A.J.D. (2010) Environmental flows and the European Water Framework Directive. *Freshwater Biology* **55,** *32-48.*

Acreman, M.C., Dunbar, M.J., Hannaford, J., Bragg, O.M., Black, A.R., Rowan, J.S. & King, J. (2006) SNIFFER WFD48 Development of Environmental Standards (Water Resources) Stage 3, 146 pp.

Acreman, M., Dunbar, M., Hannaford, J., Mountford, O., Wood, P., Holmes, N., Cowx, I.G., Noble, R., Extence, C., Aldrick, J., King, J., Black, A. & Crookall, D. (2008) Developing environmental standards for abstractions from UK rivers to implement the EU Water Framework Directive. *Hydrological Sciences Journal* **56***,* 1105-1120.

Acreman, M., Aldrick, J., Binnie, C., Black, A., Cowx, I.G., Dawson, H., Dunbar, M., Extence, C., Hannaford, J., Harby, A., Holmes, N., Jarritt, N., Old, G., Peirson, G., Webb, J.& Wood, P. (2009) Environmental flows from dams: the water framework directive. *Engineering Sustainability* **162,** 13-22**.**

Alabaster, J.S. (1990) The temperature requirements of adult Atlantic salmon, *Salmo salar* L., during their upstream migration in the river Dee. *Journal Fish Biology* **37,** 659–661.

Aldrick, J. (2010). Good practice guidance for hydropower. An Environment Agency – CIWEM conference. Hydropower opportunities, challenges and sharing best practice. June 23rd 2010. SOAS, London.

Almodovar, A. & Nicola, G.G. (1999) Effects of a small hydropower station upon brown trout *Salmo trutta* L. in the river Hoz Seca (Tagus Basin, Spain) one year after regulation. *Regulated Rivers: Research & Management* **15,** 477-484.

Amaral, S.V., Winchell F.C., McMahon B.J. & Dixon D.A. (2003) Evaluation of angled bar racks and louvers for guiding silver phase American eels. In: D.A. Dixon (ed.) *Biology, Management and Protection of Catadromous Eels.* Symposium 33. Bethesda, MD, USA: American Fisheries Society, pp. 367-376.

Anagnostopoulos, J.S. & Papantonis, D.E. (2007) Optimal sizing of run-of-river small hydropower plant. *Energy Conservation & Management* **48,** *2663-2670.*

Anderson, E.P., Freeman, M.C. & Pringle, C.M. (2006) Ecological consequences of hydropower development in Central America: Impacts of small dams and water diversion on neotropical stream fish assemblages. *River Research & Applications* **22**, 397-411.

Anderson, E.P., Pringle, C.M., & Freeman, M.C. (2008) Quantifying the effect of river fragmentation by hydropower dams in the Sarapiqui River basin, Costa Rica. *Aquatic Conservation: Marine & Freshwater Ecosystems* **18,** 408-417.

Angelaki, V., & Harbor, J.M. (1995) Impacts of flow diversion for small hydroelectric power plants on sediment transport, northwest Washington, *Physical Geog*raphy **16**, 432- 443.

Angermeier, P.L. (1987) Spatiotemporal variation in habitat selection by fishes in a small Illinois stream. In: D.C. Matthews & W.J. Heins (eds) *Community Ecology of North American Stream Fishes.* University of Oklahoma Press, Norman, pp. 52-60.

Angilletta, M.J., Steel, E.A., Bartz, K.K., Kingsolver, J.G., Scheuerell, M.D., Beckman, B.R. & Crozier, L.G. (2008) Big dams and salmon evolution: changes in thermal regimes and their potential evolutionary consequences. *Evolutionary Applications* **1**, 286-299.

Annear, T., Chisholm, I., Beecher, H., Locke, A., Aarestad, P., Coomer, C., Estes, C., Hunt, J., Jacobson, R., Jobsis, G., Kauffman, J., Marshall, J., Mayes, K., Smith, G., Wentworth, R. & Stalnaker, C. (2004) Instream Flows for Riverine Resource Stewardship (revised edn), Instream Flow Council, Cheyenne, WY.

Anon. (2008) *Scottish Hydropower Resource Study.* The Scottish Government, 57 pp.
Aprahamian, M.W., Walker, A. M., Williams, B., Bark, A. & Knights, B. (2007) On the application of models of European eel (Anguilla anguilla) production and escapement to the development of Eel Management Plans: the River Severn. *ICES Journal of Marine Science* **64,** 1472-1482.
Armstrong, G.S., Aprahamian, M.W., Fewlings, G.A., Gough, P.J., Reader, N.J. & Varallo, P.V. (2010) Environment Agency Fish Pass Manual: Guidance Notes on the Legislation, Selection and Approval Of Fish Passes in England And Wales. Bristol: Environment Agency.
Armstrong, J.D., Braithwaite, V.A. & Fox, M. (1998) The response of wild Atlantic salmon parr to acute reductions in water flow. *Journal Animal Ecology* **66**, 203 –211.
Armstrong, J.D., Kemp, P.S., Kennedy, G.J.A., Ladle, M. & Milner, H.J. (2003) Habitat requirements of Atlantic salmon and brown trout in rivers and streams. *Fisheries Research* **62,** 143-170.
Arnekleiv, J. & Kraabøl, M. (1996) Migratory behaviour of adult fast-growing brown trout (*Salmo trutta,* L.) in relation to water flow in a regulated Norwegian river. *Regulated Rivers: Research & Management* **12,** 39-49.
Arthington, A.H., Bunn, S.E., Poff, N.L. & Naiman, R.J. (2006) The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* **16,** 1311-1318.
Ausubel, J.H. (2007) Renewable and nuclear heresies. *Nuclear Governance, Economy and Ecology* **1,** 229-243.
Bagenal, T.B. (1978) *Methods for the Assessment of Fish Production in Fresh Waters*. BP Handbook No 3 (3rd edition). Oxford: Blackwell Scientific Publications.
Bagenal, T.B. & Tesch, F.W. (1978) Age and growth. In: Bagenal, T.B. (ed.) *Methods for Assessment of Fish Production in Fresh Waters.* Third Edition. Blackwell Scientific Publications, pp 101-136.
Banks, J. (1969) A review of the literature on the upstream migration of adult salmonids. *Journal of Fish Biology* **1,** 85–136.
Baran, P., Delacoste, M., Dauba, F., Lascaux, J.M., Belaud, A. & Lek, S. (1995) Effects of reduced flow on brown trout (*Salmo trutta* L.) populations downstream dams in French Pyrenees. *Regulated Rivers: Research & Management* **10**, 347–361.
Baras, E., Lambert, H. & Philippart, J.C. (1994) A comprehensive assessment of the failure of *Barbus barbus* spawning migrations through a fish pass in the canalized River Meuse (Belgium). *Aquatic Living Resources* **7,** 181-189.
Barlaup, B.T., Gabrielsen, S.E., Skoglund, H. & Wiers, T. (2008) Addition of spawning gravel – A means to restore spawning habitat of Atlantic salmon (*Salmo salar* L.) and anadromous and resident brown trout (*Salmo trutta* L.) in regulated rivers. *River Research & Applications* **24,** 543-550.
Barlaup, B.T., Lura, H., Sægrov, H. & Sundt, R.C. (1994) Inter- and intra-specific variability in female spawning behaviour. *Canadian Journal of Zoology* **72,** 636-642.
Barnard, S. & Wyatt, R. J. (1995) A guide to HABSCORE field survey methods and the completion of standard forms. Report to NRA, **No. 401**; WRc.
Baron, J.S., Poff, N.L., Angermeier, P.L., Dahm, C.N., Gleick, P.H., Hairston, N.G., Jackson, R.B., Johnston, C.A., Richter, B.D. & Steinman, A.D. (2002) Meeting ecological and societal needs for freshwater. *Ecological Applications* **12**, 1247-1260.
Bash, J., Berman, C. & Bolton, S. (2001) Effects of turbidity and suspended solids on salmonids. Center for Streamside Studies, University of Washington, Seattle, WA.
Bratrich, C., Truffer, B., Jorde, K., Markard, J., Meier, W., Peter, A., Schneider, M. & Wehrli, B. (2004) Green Hydropower: a New Assessment Procedure for River Management. *River Research and Applications* **20**, 865-882.
Baumgartner, L.J. (2007) Diet and feeding habits of predatory fishes upstream and downstream of a low-level weir. *Journal Fish Biology* **70,** 879-894.
Baxter, G. (1961) River utilization and the preservation of migratory fish life. *Proceedings of the Institute of Civil Engineers* **18,** 225-244.
Bean, C & Thin, F. (2008) Hydroelectricity – Impacts and opportunities for the natural heritage. Energy and the Natural Heritage, pp. 155-184.
Becker, J.M., Abernethy, C.S. & Dauble, D.D. (2003) Identifying the effects on fish of changes in water pressure during turbine passage. Hydro review, pages 1-5.
Behrmann-Godel, J. & Eckmann, R. (2003) A preliminary telemetry study of the migration of silver European eel (*Anguilla anguilla*) in the river Mosel, Germany. *Ecology of Freshwater Fish* **12**, 196-202.

Bell, M. & DeLacy, A. (1972) A compendium on the survival of fish passing through spillways and conduits. Fisheries engineering research program, US army engineers division, North Pacific corps of engineers, Portland, Oregon.

Belpaire, C.G.J., Goemans, G., Geeraerts, C., Quataert, P., parmentier, K., Hagel, P. & De Boer, J. (2009) Decreasing eels stock: survival of the fattest? *Ecology of Freshwater Fish* **18,** 197-214.

Benstead, J., March, J., Pringle, C. & Scatena, F.N. (1999) Effects of low-head dam and water abstraction on migratory tropical biota. *Ecological Applications* **9,** 656-668.

Berg, O.K. & Moen, V. (1999) Inter and intrapopulation variation in temperature sum requirements at hatching in Norweigian Atlantic salmon. *Journal Fish Biology* **54,** 636-647.

Berg, O.K., Hendry, A.P., Svendsen, H.B., Bech, C., Arnekleiv, J.V. & Lohrmann, A. (2001) Maternal provisioning of offspring and the use of those resources during ontogeny: variation within and between Atlantic salmon families. *Functional ecology* **15,** 13-23.

Bernies, D., Brinker, A. & Daugschies, A. (2011) An invasion record for the swimbladder nematode *Anguillicoloides crassus* in European eel *anguilla anguilla* in a deep warm-monomictic (corrected) lake, from invasion to steady state. *Journal of Fish Biology* **79(3),** 726-746.

Beschta, R. L. & Jackson, W.L. (1979) The intrusion of fine sediments into a stable gravel bed, *Fisheries Research Board of Canada* ***36*****,** 204-210.

BHA (2008) BHA news. Hydropower – the pioneer of renewable energy. Stepping into the spotlight. The Institute of Fisheries Management oversteps the mark. Issue 3.

BHA (2011) BHA spotlight. Hydropower – the pioneer of renewable energy. The gold rush north and south of the border. Issue 1.

Biggs, B.J.F., Nikora, V.I. & Snelder T.H. (2005) Linking scales of flow variability to lotic ecosystem structure and function. *River Research & Applications* **21**, 283-298.

Bjorn, T.C. & Perry, C.A. (1992) A Review of Literature Related to Movements of Adult Salmon and Steelhead Past Dams and Through Reservoirs in the Lower Snake River, U.S. Army Corps of Engineers, North Pacific Division, Technical Report 92-1, Portland, OR.

Bolland, J.D., Cowx, I.G. & Lucas, M.C. (2008) Movements and habitat use of wild and stocked juvenile chub, *Leuciscus cephalus* (L.), in a small lowland river. *Fisheries Management and Ecology* **15**, 401-407.

Bonhommeau, S., Castonguay, M., Rivot, E., Sabatie, R. & Le Pape, O. (2010) The duration of migration of Atlantic *Anguilla* larvae. *Fish and Fisheries* **11,** 289-306.

Boubee, J.A.T & Williams, E.K. (2006) Downstream passage of silver eels at a small hydroelectric facility. *Fisheries Management and Ecology* **13**, 165-176.

Boubee, J., Jellyman, D. & Sinclair, C. (2008) Eel protection measures within the Manapouri hydro-electric power scheme, South Island, New Zealand. *Hydrobiologia* **609**, 71-82.

Bradford, M. J. (1997) An experimental study of stranding of juvenile salmonids on gravel bars and in side channels during rapid flow fluctuations. *Regulated Rivers: Research & Management* **13,** 395-401.

Brasher, A. (2003) Impacts of human disturbances on biotic communities in Hawaiian streams. *BioScience* **53**,1052–1060.

Bratrich, C., Truffer, B., Jorde, K., Markard, J., Meier, W., Peter, A., Schneider, M. & Wehrli, B. (2004) Green hydropower: A new assessment procedure for river management. *River Research & Applications* **20**, 865-882.

Britton, J. R. (2003) Fisheries Monitoring Programme Work Instruction 3.3: Sampling fish scales for age and growth determination. Environment Agency Management System.

Brisbane Declaration, (2007) Available at: http: / / www.nature.org/initiatives/freshwater/files/brisbane_declaration_with_organisations_final.pdf.

Bruno, G.S. & Fried, L. (2008) Focus on small hydro. *Renewable Energy Focus* **9**, 54-57.

Bunn, S.E. & Arthington A.H. (2002) Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* **30**, 492–507.

Bunt, C.M., Cooke, S.J., Katopodis, C. & McKinley, R.S. (1999) Movement and summer habitat of brown trout (*Salmo trutta*) below a pulsed discharge hydroelectric generating station. *Regulated Rivers-Research & Management* **15**, 395-403.

Cada, G.F. (1998) Better science supports fish-friendly turbine designs. *Hydro Review*, Vol. XVII, No. 6, November 1998. pp 52-61.

Cada, G.F. (2001) The Development of Advanced Hydroelectric Turbines to Improve Fish Passage Survival. Bioengineering Feature. Volume 26 No 9, Pp 14-23.

Cada, G., Loar, J., Garrison, L., Fisher, J. & Neitzel, D. (2006) Efforts to reduce mortality to hydroelectric turbine - passed fish: Locating and quantifying damaging shear stresses. *Environmental Management* **37**, 898-906.
Calles, E.O. & Greenberg, L.A. (2005) Evaluation of nature-like fishways for re-establishing connectivity in fragmented salmonid populations in the River Eman. *River Research and Applications* **21**, 951–960.
Calles, E.O. & Greenberg, L.A. (2007) The use of two nature-like fishways by some fish species in the Swedish River Eman. *Ecology of Freshwater Fish* **16**, 183-190.
Campbell, J.S. (1977) Spawning characteristics of brown trout and sea trout *Salmo trutta* L. in Kirk Burn, River Tweed, Scotland. *Fish Biology* **11,** 217-229.
Campbell, D.C. (1985) “Need” for small hydro in an environmental context: three case studies”. *cx*In: *Symposium on Small Hydropower and Fisheries*. Denver., Colorado, May 1985. pp. 121-126.
Carle, F.L. & Strub, M.R. (1978) A new method for estimating population size from removal data *Biometrics* **34**, 621-630.
Celtic Sea trout project (2010) Manual on Sea Trout Ageing, Digital Scale Reading and Growth Methodology. 49pp.
Cereghino, R. & Lavandier, P. (1998) Influence of hypolimnetic hydropeaking on the distribution and population dynamics of Ephemeroptera in a mountain stream. *Freshwater Biology* **40**, 385–399.
Chadwick, S., Kinghts, b., Throley, J.L. & Bark, A. (2007) A long term study of population charachteristics and downstream migrations of the European eel *Anguilla Anguilla* (L.) and the effects of a migration barrier in the Girnock Burn, north-east Scotland. *Fish Biology* **70,** 1535-1553.
Clay, C. (1995) *Design of Fishways and Other Fish Facilities*. Boca Raton: CRC Press, Inc.
Copeman, V.A. (1997) The impact of micro hydropower on the aquatic environment. *Water and Environment* **11,** 431-435.
Copestake, P. (2005) Hydropower and environmental regulation - A Scottish perspective. Proceedings of the Annual Spring Conference of the British-Ornithologists-Union, Leicester, England, Apr 01-03, 2005,169-179.
Corbacho, C. & Sanchez, J.M. (2001) Patterns of species richness and introduced species in native freshwater fish faunas of a Mediterranean-type basin: the Guadiana River (southwest Iberian Peninsula). *Regulated Rivers: Research & Management* **17,** 699–707.
Cortes, R.M.V., Ferreira, M.T., Oliveira, S.V. & Oliveira, D. 2002. Macroinvertebrate community structure in a regulated river segment with different flow conditions. *River Research and Applications* **18,** *367-382.*
Covich, A.P.,Crowl, T.A. & Scatena, F.N. (2003) Effects of extreme low flows on freshwater shrimps in a perennial tropical stream. *Freshwater Biology* **48**, 1199–1206.
Cowx, I.G. (1996) The integration of fish stock assessment into fisheries management. In: Cowx I.G. (ed.) *Stock Assessment in Inland Fisheries*. Fishing News Books, Blackwell Science, Oxford, pp. 495-506.
Cowx, I.G. (1998) Review of fish passage facilities in the UK: Issues and options for future development In: M. Jungwith, S. Schmutz & S. Weiss (eds) *Fish Migration and Fish bypass Channels*. Oxford: Fishing News Books, Blackwell Science. pp. 220-235.
Cowx, I.G. & Lamarque, P. (eds) (1990) *Fishing with Electricity*, Fishing News Books Ltd, Blackwells Scientific Publications Oxford. 245 pp.
Cowx, I.G. & Noble, R.A.A. (2004). Flow and level criteria for coarse fish and conservation species: Phase 1. Report to Environment Agency, 80 pp.
Cowx, I.G. & Welcomme, R.L. (eds) (1998) *Rehabilitation of Rivers for Fish. Oxford:* FAO and Fishing News Books, Blackwell Science, 260 pp.
Cowx, I.G., Harvey, J.P., Noble, R.A. & Nunn, A.D. (2009) Establishing survey and monitoring protocols for the assessment of conservation status of fish populations in river Special Areas of Conservation in the UK. *Aquatic Conservation: Marine and Freshwater Ecosystems* **19**, 96-103.
Cowx, I.G., Axford, S.A., Robson, A. & Harvey, J.P. (2012) Impact of run-of-river hydro-schemes upon fish populations: Comparison of UK Regulatory Frameworks. Guidelines for run-of-river hydropower development SNIFFER Project WFD114 Phase 1 Draft Report.
Crisp, D.T. (1993) The ability of UK salmonid alevins to emerge through a sand layer. *Journal of Fish Biology* **43,** 656-658.

Crisp, D.T. (2000) *Trout and Salmon. Ecology, Conservation and Rehabilitation.* Oxford: Blackwell Science, pp. 16-22.
Crocker, M (2010). Fisheries, biodiversity and hydropower issues. Environment Agency. CIWEM conference outputs. Hydropower opportunities, challenges and sharing best practice. 23/06/2010. SOAS, London.
Crozier, W. W. & Kennedy, G. J. A. (2001) Relationship between freshwater angling catch of Atlantic salmon and stock size in the River Bush, Northern Ireland. *Journal of Fish Biology* **5**, 240-247
Crozier, W.W., Schon, P.J., Chaput, G., Potter, E.C.C., Maioleidigh, N.O. & MacLean, J.C. (2004) Managing Atlantic salmon (S*almo salar* L.) in the mixed stock environment: challenges and considerations. *ICES Journal of Marine Science* **61**, 1344-1358.
Davey, A.J.H. & Kelly D.J. (2007) Fish community responses to drying disturbances in an intermittent stream: a landscape perspective. *Freshwater Biology* **52**, 1719–1733.
Davey, A.J.H., Kelly, D.J. & Biggs, B.J.F. (2006) Refuge-use strategies of stream fishes in response to extreme low flows. *Journal of Fish Biology* **69**, 1047-1059.
DECC (2010) Department of Energy and Climate Change. Digest of United Kingdom energy statistics 2010: long term trends. pp 155-221. http://www.decc.gov.uk/assets/decc/statistics/publications/dukes/324-dukes-2010-longterm.pdf.
De gaudemar, B. & Beall, E. (1998) Effects of overripening on spawning behaviour and reproductive success of Atlantic salmon females spawning in a controlled flow channel. *Journal of Fish Biology* **53,** 434-446.
Dekker, W. (2000) The fractal geometry of the European eel stock*. ICES Journal of Marine Science* **57**, 109–121.
Dekker, W. (2003) Did lack of spawners cause the collapse of the European eel, *Anguilla anguilla*? *Fisheries Management and Ecology* **10**, 365–376.
De Leaniz, C.G. (2008) Weir removal in salmonid streams: implications, challenges and practicalities. *Hydrobiologia* **609,** 83-96.
Demirbas, A. (2007) Focus on the world: status and future of hydropower. Energy sources part B: *Economics Planning and Policy* **2**, 237-242.
De Vocht, A & Baras, E. (2003) Effect of hydro-peaking on migrations and home range of adult Barbel (*Barbus barbus*) in the River Meuse. *Aquatic telemetry: advances and applications. Proceedings of the Fifth Conference on Fish Telemetry held in Europe. Ustica, pp.* 35-44.
Downes, B.J., Barmuta, L.A., Fairweather, P.G., Faith, D.P., Keough, M.J., Lake, P.S., Mapstone, B.D. & Quin, G.P. (2002) Monitoring ecological impacts:concepts and practice in flowing waters. Cambridge University Press, Cambridge, UK.
Drinkwater, K. F. & Frank, K.T. (1994) Effects of river regulation and diversion on marine fish and invertebrates. *Aquatic Conservation* **4,** 135–151.
Dudgeon, D., Arthington, A.H., Gessner M.O., Kawabata, Z., Knowler, D.J., Leveque, C., Naiman, R.J., Prieur-Richard, A., Soto, D., Stiassny, M.L.J. & Sullivan, C.A. (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* **81**, 163–182.
Dynesius, M. & Nilsson, C. (1994) Fragmentation and flow regulation or river systems in the northern third of the world. *Science* **266,** 753-762.
Dytham, C. (2009) Choosing and using statistics: a biologist's guide - Malden, MA. Blackwell 2009. Second edition.
Eberstaller, J., Hinterhofer, M. & Parasiewicz, P. (1998) The effectiveness of two nature-like bypass channels in an upland Austrian river. In: M. Jungwirth, S. Schmutz. & S. Weiss (eds) *Fish Migration and Fish Bypasses*. Oxford: Fishing News Books, Blackwell Publications, pp. 363–383.
Egre, D & Milewski, J.C. (2002) The diversity of hydropower projects. *Energy Policy* **30**, 1225-1230.
Elliott, J.M. (1986) Spatial distribution and behavioural movements of migratory trout, *Salmo truttu,* in a Lake District stream. *Journal of Animal Ecology* **55**, 907-922.
Elliott, J.M., Hurley, M.A. & Elliott, J.A. (1997) Variable effects of droughts on the density of a sea-trout *Salmo trutta* population over 30 years. *Applied Ecology* **34**, 1229–1238.
Enders, E.C., Scruton, D.A. & Clarke, K.D. (2009) The Natural Flow Paradigm and Atlantic salmon - moving from concept to practice. *River Research & Applications* **25**, 2-15.
Environment Agency (2005) Screening for intakes and outfalls: a best practice guide. Environment Agency Report. 154 pp.

Environment Agency (2009a). Good practice guidelines to the Environment Agency hydropower handbook. The environmental assessment of proposed low head hydropower developments: Environment Agency Report, 45 pp.

Environment Agency (2009b) Consultation – Part 2 new sections. “Good practice guidelines to the environment agency handbook – The environmental assessment of proposed low head hydropower developments. Environment Agency report, 7pp.

Environment Agency (2010a). Opportunity and environmental mapping sensitivity mapping for hydropower in England and Wales. Non-technical report. 26pp.

Environment Agency (2010b) Streamlining permitting of hydropower projects in England and Wales. Consultation document. Enironment Agency report, 41pp.

Environment Agency (2010c) Hydropower permitting review. Summary of consultation responses December 2010. Environment Agency report, 23pp.

European Commission Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.

Everitt, R.R. (1992) Environmental Effects Monitoring Manual. Prepared for the Federal Environmental Assessment Review Office and Environment Canada, Environmental Assessment Division, Inland Waters Directorate. Ottawa, ON.

Felley, J.D. & Felley, S.M. (1987) Relationships between habitat selection by individuals of a species and patterns of habitat segregation among species: fishes of the Calcasieu drainage. In: D.C. Matthews & W.J. Heins (eds) *Community Ecology of North American Stream Fishes*. University of Oklahoma Press, Norman, pp. 61–68.

Fette, M., Weber, C., Peter, A. & Wehrli, B. (2007) Hydropower production and river rehabilitation: A case study on the Alpine river. *Environmental Modelling and Assessment* **12**, 257-267.

Fishtek Consulting (2009) The archimedes screw turbine. Assessment of three leading edge profiles. 8mm steel edge. Hard rubber. Compressible bumper. 25pp.

Fjellheim, A. & Raddum, G.G. (1996) Weir building in a regulated west Norwegian River: Long-term dynamics of invertebrates and fish. *Regulated Rivers-Research & Management* **12**, 501-508.

Fleming, I.A. (1996) Reproductive strategies of Atlantic salmon: ecology and evolution. *Reviews in Fish Biology and Fisheries* **6**, 379–416.

Floodmark, L.E.W., Vollestad, L.A. & Forseth, T. (2004) Performance of juvenile brown trout exposed to fluctuating water level and temperature. *Journal of Fish Biology* **65,** 460-470.

Fontaine, Y.A., Pisam, M., Moal, C.L. & Rambourg, A. (1995) Silvering and gill ‘mitochondria-rich’ cells in the eel, *Anguilla anguilla. Cell Tissue Res*. **281**, 465-471.

Forrest, N., Abell, T., Baker, K., Robertson, K., Duncan, N., Hawkins, S., Baldock, A., Whetter, B. & Hieatt, M. (2008) Scottish Hydropower Resource Study. Forum for Renewable Energy Developments in Scotland (FREDS), Final report to Scottish Government, 57 pp.

Fraenkel, P., Paish, O., Bokalders V., Harvey A., Brown A. & Edwards R. (1991) *Hydropower: a guide for development workers*. London: IT Publications Ltd, 138 pp.

Francis, R.I.C.C. (1990) Back-calculation of fish length: A critical review. *Journal of Fish Biology* **36**, 883-902.

Freyhof, J. & Brooks, E. (2011) European red list of freshwater fishes. Publications Office of the European Union, Luxembourg.

Frissell, C.A. (1993) Topology of extinction and endangerment of native fishes in the Pacific Northwest and California U.S.A. *Conservation Biology* **7,** 342–354.

Furniss, R. (2010) The voice of angling. Angling Trust. CIWEM conference outputs. Hydropower: opportunities, challenges and sharing best practice. 23/06/2010. SOAS, London.

Galat, D.L. & Lipkin, R. (2000) Restoring ecological integrity of great rivers: historical hydrographs aid in defining reference conditions for the Missouri River. *Hydrobiologia* **422/423,** 29–48.

Gehrke, P.C., Brown, P., Schiller, C.B., Moffatt, D.B. & Bruce, A.M. (1995) River regulation and fish communities in the Murray-Darling River System, Australia. *Regulated Rivers: Research & Management* **11,** 363–375.

Gehrke, P.C., Gilligan, D.M. & Barwick, M. (2002) Changes in fish communities of the Shoalhaven River 20 years after construction of Tallowa Dam, Australia. *River Research Applications* **18**, 265–286.

Gerlier, M. & Roche, P. (1998) A radio telemetry study of the migration of Atlantic salmon *Salmo salar* L. and sea trout *Salmo trutta trutta* L. in the upper Rhine. *Hydrobiologia* **371/372,** 283–293.

Gibbins, C., Shellberg, J., Moir, H., & Soulsby, C. (2008) Hydrological influences on adult salmonid migration, spawning, and embryo survival. *American Fisheries Society Symposium* **65,** 195-223.

Gilvear, D.J., Heal, K.V. & Stephen, A. (2002) Hydrology and the ecological quality of Scottish river ecosystems. *The Science of the Total Environment* **294**, 131-159.

Giorgi, A.E., Swan, G.A & Zaugg, W.S. (1988) Susceptibility of Chinook salmon smolts to bypass systems at hydroelectric dams. *North American Journal of Fisheries management* **8**, 25-29.

Giske, J., Huse, G. & Fiksen, O. (1998) Modelling spatial dynamics of fish. *Reviews in Fish Biology and Fisheries* **8,** 57–91.

Gladwell M. (2000) *The Tipping Point: How Little Things Can Make a Big Difference.* Boston, MA: Little Brown and Company.

Godfrey, J.D. (2005) Site condition monitoring of Atlantic Salmon SACs. Report to Scottish Natural Heritage.

Godinho, A.L. & Kynard, B. (2009) Migratory fishes of Brazil: Life history and fish passage needs. *River Research & Applications* **25,** *702-712.*

Godinho, A.L., Kynard, B. & Godinho, H.P. (2007) Migration and spawning of female surubim (*Pseudoplatystoma corruscans*, Pimelodidae) in the Sao Francisco River, Brazil. *Environmental Biology of Fishes* **80,** 421–433.

Gollock, M. J., Kennedy, C.R. & Brown, J.A. (2005) European eels Anguilla anguilla (L.), infected with Anguillicola crassus exhibit a more pronounced stress response to severe hypoxia than uninfected eels. *Journal of Fish Diseases* **28,** 429-436.

Goodwin, R. A., Nestler, J.M., Anderson, J.J., Weber, L.J. & Loucks, D.P. (2006) Forecasting 3-D fish movement behaviour using a Eulerian-Lagrangian-agent method (ELAM). *Ecological Modelling* **192**, 197-223.

Gore, J.A., Hamilton, S.W. (1996) Comparison of flow-related habitat evaluations downstream of low-head weirs on small and large fluvial ecosystems. *Regulated Rivers: Research & Management* **12**, 459-469.

Gorman, O.T. & Karr, J.R. (1978) Habitat structure and stream fish communities. *Ecology* **59,** 507–515.

Gosset, C., Travade, F., Durif, C, Rives, J. & Elie, P. (2005) Tests of two types of bypass for downstream migration of eels at a small hydroelectric power plant. *River Research & Applications* **21,** 1095-1105.

Gosset, C., Rivers, J. & Labonne, J. (2006) Effect of habitat fragmentation on spawning migration of brown trout (*Salmo trutta* L.). *Ecology of Freshwater Fish* **15**, 247-254.

Gowans, A.R.D., Armstrong, J.D. & Priede, I.G. (1999) Movements of adult Atlantic salmon through a reservoir above a hydroelectric dam: Loch Faskally. *Journal of Fish Biology* **54,** 727–740.

Gowans, A.R.D., Armstrong, J.D., Priede, I.G. & McKelvey, S. (2003) Movements of Atlantic salmon migrating upstream through a fish-pass complex in Scotland. *Ecology of Freshwater Fish* **12,** 177-189.

Grassi, G.B. (1896) The reproduction and metamorphosis of the common eel (Anguilla vulgaris). Proceedings of the Royal Society of London **60**, 260–271.

Growns, I. (2008) The influence of changes to river hydrology on freshwater fish in regulated rivers of the Murray-Darling basin. *Hydrobiologia* **596,** 203-211.

Guiny, E., Armstrong, J.D. & Ervine, D.A. (2003) Preferences of mature male brown trout and Atlantic salmon parr for orifice and weir fish pass entrances matched for peak velocities and turbulence. *Ecology of Freshwater Fish* **12**, 190-195.

Habit, E., Belk, M.C. & Parra, O. (2007) Response of the riverine fish community to the construction and operation of a diversion hydropower plant in central Chile. *Aquatic Conservation* **17**, 37-49.

Hadderingh, R.H. & Bakker, H.D. (1998) Fish mortality due to passage through hydroelectric power stations on the Meuse and Vecht rivers. In: M. Jungwirth, S. Schmutz. & S. Weiss (eds) *Fish Migration and Fish Bypasses*. Oxford: Fishing News Books, Blackwell Publications, pp. 315-328.

Hakala, J.P. & Hartman, K.J. (2004) Drought effect on stream morphology and brook trout (*Salvelinus fontinalis*) populations in forested headwater streams. *Hydrobiologia* **515**, 203–213.

Halls, A & Kshatriya, M. (2009) Modelling the cumulative effects of mainstream hydropower dams on migratory fish populations in the lower Mekong basin. MRC Technical Paper No. 25, Mekong River commission, Vientiane, 101 pp.

Haro, A., Castro-Santos, T. & Boubee, J. (2000) Behaviour and passage of silver-phase American eels, *Anguilla rostrata* (LeSueur), at a small hydroelectric facility. *Dana* **12**, 33-42.

Harris J.H. & Mallen-Cooper M. (1994) Fish passage development in the rehabilitation of fisheries in mainland south-eastern Australia. In: I.G. Cowx (ed.) Rehabilitation of Freshwater Fisheries. Oxford: Fishing News Books, pp. 185–193.

Hart, D.D., Johnson, T.E., Bushaw-Newton, K.L., Horwitz, R.J., Bednarek, A.T., Charles, D.F., Kreeger, D.A. & Velinsky, D.J. (2002) Dam removal: Challenges and opportunities for ecological research and river restoration. *BioScience* **52,** 669–681.

Hartvich, P., Dvorak, P., Tlusty, P & Vrana, P. (2008) Rotation screen prevents fish damage in hydroelectric power stations. *Hydrobiologia* **609**, 163-176.

Hecker, G.E. & Cook, T.C. (2005) Development and evaluation of a new helical fish-friendly hydroturbine. *Hydrological Energy* **131**, 835–844.

Heggeberget, T.G. (1988) Timing of spawning of Norwegian Atlantic salmon (*Salmo salar*). *Canadian Journal of Fisheries & Aquatic Science* **45,** 845–849.

Heggenes, J. (1996) Habitat selection by brown trout (S*almo trutta)* and young Atlantic salmon (*S.salar*) in streams: static and dynamic hydraulic modelling. *Regulated River: Research and Management* **12,** 155-169.

Heggenes, J**.** (2002) Flexible summer habitat selection by wild, allopatric brown trout in lotic environments. *Trans. Am. Fish. Soc.* **131**, 287-298.

Heinz Center, (2002) *Dam Removal. Science and Decision Making.* Washington: The Heinz Center.

Hirzinger, V., Keckeis, H., Nemeschkal, H.L. & Schiemer, F. (2004) The importance of inshore areas for adult fish distribution along a free-flowing section of the Danube, Austria. *River Research & Applications* **20,** 137–149.

Hogan, S. (2005) Run off river micro-hydropower: a solution to the Banff corridors electrical capacity dilemma. www.utsb.ca/PDFs/micro_hydro.pdf.

Hunt, R.L. (1976) A long-term evaluation of trout habitat development and its relation to improving management related research, *Transactions of American Fisheries Society* **105**, 361–364.

Hvidsten, N.A. (1985) Mortality of pre-smolt Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., caused by fluctuating water levels in the regulated River Nidelva, central Norway. *Journal of Fish Biology* **27**, 711–718.

Institution of Engineering & Technology. (2007) Hydroelectric Power. 11pp.

Irish Fisheries Board. (2007) The Central and Regional Fisheries Boards of Ireland guidelines for hydropower development. Guidelines on the planning, design, construction and operation of small-scale hydro-electric schemes and fisheries. Fisheries Ireland. Our Natural Heritage. 54pp.

IUCN (2008) *IUCN Red List of Threatened Species*. Available at www.iucn.org/knowledge/ red list.

Jager, H.I., Bevelhimer, M.S. (2007) How run-of-river operation affects hydropower generation and value. *Environmental Management* **40**, 1004-1015.

Jager, H.I. & Smith, B.T. (2008) Sustainable reservoir operation: Can we generate hydropower and preserve ecosystem values?" *River Research & Applications* **24**, 340–352.

Jansen, H.M., Winter, H.V., Bruijs, M.C.M. & Polman, H.J.G. (2007) Just go with the flow? Route selection and mortality during downstream migration of silver eels in relation to river discharge. *ICES Journal of Marine Science* **64,** 1437-1443.

Jansson, R. (2002) The biological cost of hydropower. Landscape ecology group. Department of Ecology and Environmental Science. Umea university.

Jansson, R., Nilsson, C. & Renofalt, B. (2000) Fragmentation of riparian floras in rivers with multiple dams. *Ecology* **81**, 899-903.

Jensen, A. J., Johnsen, B. O. & Hansen, L. P. (1989). Effect of river flow and water temperature on the upstream migration of adult Atlantic salmon *Salmo salar* L. In the River Vefsna, northern Norway. In *Salmonid Migration and Distribution Symposium* (Brannon, E. & Jonsson, B., eds), pp. 140–146. Seattle: School of Fisheries, University of Washington.

Jonsson, B. (1989) Life history and habitat use of Norwegian brown trout (Salmon trutta). *Freshwater Biology* **21** 71-86.

Jonsson, B. & Jonsson, N. (2002) Migration of anadromous brown trout *Salmo trutta* in a Norwegian river. *Freshwater Biology* **47,** 1391–1401.

Jonsson, B & Jonsson, N. (2011) Ecology of Atlantic Salmon and Brown Trout. Habitat as a template for life histories. Fish and Fisheries Series 33. Springer Science + Business media.

Jonsson, B. & Waples, R. (1999) Convenors-Theme session on diadromous fish extinction: threats on local and global scales, 1997. *ICES Journal of Marine Science* **56**, 405–472.

Jonsson N., Jonsson, B., Hansen, L.P. & Aass, P. (1994) Effects of sea water acclimatization and release sites on survival of hatchery reared brown trout *Salmo trutta. Journal of Fish Biology* **44,** 973-981.

Jonsson, B., Jonsson, N. & Hansen, L.P. (1990a) Does juvenile experience affect migration and spawning of adult Atlantic Salmon*. Behavioural Ecology and Social biology **22,** 225-230.*

Jonsson, B., Jonsson, N. & Hansen, L.P. (1990b) Partial segregation in the timing of migration of Atlantic Salmon of different ages*. Animal Behaviour **40,** 313-321.*

Jonsson, B., Jonsson, N. & Hansen, L.P. (2007) Factors affecting river entry of adult Atlantic salmon in a small river. *Journal Fish Biology* **71,** 943-956.

Jonsson, N. (1991*).* Influence of water flow, water temperature and light on fish migration in rivers*. Nordic Journal of Freshwater Research* **66***,* 20–35*.*

Jonsson, N. & Jonsson, N. (2006) Life histories of anadromous brown trout. In: Harris, G, Milner, N (eds) Sea trout: biology, conservation and management. Blackwell, Oxford.

Jowett, I. G. (1997) Instream flow methods: A comparison of approaches. *Regulated Rivers-Research & Management* **13**, 115-127.

Jowett, I.G., Richardson, J. & Bonnett, M. L. (2005) Relationship between flow regime and fish abundances in a gravel-bed river, New Zealand. *Journal of Fish Biology* **66**, 1419–1436.

Jungwirth, M. (1996) Bypass Channels at weirs as appropriate aids for fish migration in Rhithral rivers. *Regulated Rivers: Research & Management* **12**, 483-492.

Jungwirth, M. (1998) River continum and fish migration-going beyond the longitudinal river corridor in understanding ecological integrity. In: M. Jungwirth, S. Schmutz. & S. Weiss (eds) *Fish Migration and Fish Bypasses*. Oxford: Fishing News Books, Blackwell Publications, pp. 19-32.

Jungwirth, M., Schmutz, S. & Weiss, S. (1998). Fish Migration and Fish Bypasses. Oxford:Blackwell Science.

Karppinen, P., Makinen, T.S., Erkinaro, J., Kostin, V.V., Sadkovskij, R.V., Lupandin, A.I. & Kaukoranta, M. (2002) Migratory and route seeking behaviour of ascending Atlantic salmon in the regulated River Tuloma*. Hydrobiologia* **483,** 23-30.

Kataria, M. (2009) Willingness to pay for environmental improvements in hydropower regulated rivers. *Energy Economics* **31**, 69-76.

Katopodis, C. (1990) Advancing the art of engineering fishways for upstream migrants. In The Proceedings of the Fifth International Symposium on Fishways ’90, Gifu, Japan, 8–10 October; 19–28.

Keaton, M., Haney, D. & Andersen, C.B. (2005) Impact of drought upon fish assemblage structure in two South Carolina Piedmont streams. *Hydrobiologia* **545**, 209–223.

Keefer, M.L., Peery, C.A., Jepson, M.A. & Stuehrenberg, L.C. (2004) Upstream migration rates of radio tagged adult Chinook salmon in the riverine habitats of the Columbia river basin. *Journal* of F*ish Biology* **65,** 1126-1141.

Kemp, P.S., Gessel, M.H. & Williams, J.G. (2008) Response of downstream migrant juvenile Pacific salmonids to accelerating flow and overhead cover. *Hydrobiologia* **609,** 205-217.

Kemp. P.S., Russon, I.J., Vowles, A.S. & Lucas, M.C. (2010) The influence of discharge and temperature on the ability of upstream migrant adult river lamprey (*Lampetra fluviatilis*) to pass experimental overshot and undershot weirs. *River Research & Applications* (in press).

Kemp, P., Sear, D., Collins, A., Naden, P. & Jones, I. (2011) The impacts of fine sediment on riverine fish. *Hydrological Processes **25***, 1800-1821.

Kennard, M.J., Pusey, B.J., Olden, J.D., Mackay, S.J., Stein, J.L. & Marsh, N. (2010) Classification of natural flow regimes in Australia to support environmental flow management. *Freshwater Biology* **55**, 171-193.

Kibel, P. (2007) Fish monitoring and live fish trials. Ritz Atro Archimedes Screw Turbine, River Dart. Phase 1 Report: Live fish trials, smolts, leading edge assessment, disorientation study, outflow monitoring. *Fishtek Consulting.*

Kibel, P. (2008) Archimedes screw turbine fisheries assessment. Phase II: Eels and Kelts. *Fishtek Consulting*.

Kibel, P., Pike, R. & Coe, T. (2009) Archimedes Leading edge assessment. *Fishtek Consulting*.

Kingsford, R.T. (2000) Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Australia Ecology* **25**, 109-127.
Kirk, R. S., Lewis, J.W. & Kennedy, C.R. (2000) Survival and transmission of *Anguillicola crassus* Kuwahara, Niimi & Itagaki, 1974 (Nematoda) in seawater eels. *Parasitology* **120,** 289–295
Kirk, R. S. (2003) The impact of *Anguillicola crassus* on European eels. *Fisheries Management and Ecology* **10,** 385-394
Klemetson, A., Amundsen, P.A. & Dempson, J.B., Jonsson, B., Jonsson, N., O'Connell, M.F. & Mortensen, E. (2003) Atlantic salmon *Salmo salar* L., brown trout *Salmo trutta* L. and Atlantic charr *Salvelinus alpines* (L.) a review of aspects of their life histories. *Ecology Freshwater Fish* **12,** 1-59.
Knaepkens, G., Baekelandt, K. & Eens, M. (2005) Fish pass effectiveness for bullhead (*Cottus gobio*), perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*) in a regulated lowland river. *Ecology of Freshwater Fish* **15,** 20–29.
Koel, T.M. (2004) Spatial variation in fish species richness of the Upper Mississippi River System. *Transactions of the American Fisheries Society* **133,** 984–1003.
Kondolf, G.M., (2000) Assessing salmonid spawning gravel quality. *Transactions of the American Fisheries Society* **129,** 262–281.
Kraft, M.E. (1972) Effects of controlled flow reduction on a trout stream. *Journal Fisheries Research Board of Canada* **29,** 1405–1411.
Kubecka, J., Matena, J. & Hartvich, P (1997) Adverse ecological effects of small hydropower stations in the Czech Republic.1. Bypass plants. *Regulated Rivers: Research & Management* **13,** 101-113.
Kynard, B. (1993) Fish behavior important for fish passage. In Fish Passage Policy and Technology, Bates K (ed.). American Fisheries Society:Bethesda; 129–134.
Kynard, B. (2004) Review of migration, research methods, and passage for downstream migrant fishes in the Northeast USA, In: M. Lintermans & B. Philips (eds) Downstream movements of fish in the Murray-Darling Basin. Proceedings of a workshop held in Canberra, June 2003; 15–23.
Labonne, J. & Gaudin, P. (2005) Exploring population dynamics patterns in *Zingel asper*: capture–mark–release methods for rare fish demography. *Conservation Biology* **19,** 463–472.
Ladle, M. (2002) *Review of Flow Needs for Fish and Fisheries*. R&D Technical report W159, Environment Agency, 99 pp.
Laine, A., Jokivirta, T. & Katapodis, C. (2002) Atlantic salmon *Salmo salar* L. And sea trout *Salmo trutta* L., passage in a regulated northern river – fishway efficiency , fish entrance and environmental factors. *Fisheries Management and Ecology* **9,** 65-77.
Lake, P.S. (2003) Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* **48**, 1161–1172.
Larinier, M. (1998) Upstream and downstream fish passage experience in France. In: M. Jungwirth, S. Schmutz. & S. Weiss (eds) *Fish Migration and Fish Bypasses*. Oxford: Fishing News Books, Blackwell Publications, pp. 127-145.
Larinier, M. (2000) Dams and Fish Migration. World Commission on Dams, Toulouse, France.
Larinier, M. (2001) Environmental issues, dams and fish migration. In: G.Marmulla, (ed.) Dams, Fish and Fisheries. Opportunities, Challenges and Conflict Resolution. *FAO Fisheries Technical Paper* **419**, 45–89.
Larinier, M. (2008) Fish passage experience at small scale hydro electric power plants in France. *Hydrobiologia* **609**, 97-108.
Larinier, M & Travade, F. (1999) The development and evaluation of downstream bypasses for juvenille salmonids at small scale hydroelectric plants in France. In: M. Odeh (ed.) *Fish Passage Technology.* American Fisheries Society, Bethesda, Maryland, USA, pp. 25-42.
Larinier, M. & Travade, F. (2002) Downstream migration: problems and facilities. *Bulletin Français de la Pêche et de la Pisciculture* **364** (Suppl.), 181–207.
LeClerc, M. (2008) *Getting down with VLH turbine R&D.* International Water Power and Dam Construction, May 2008.
Levin, P. S. & Tolimieri, N. (2001) Differences in the impacts of dams on the dynamics of salmon populations. *Animal Conservation* **4,** 291–299.
Lichatowhich, J & Cramer, S. (1979) Parameter selection and sample size in studies of anadromous salmonids. Oregon department of fish and wildlife, Information Report Series, Fisheries number 80-1, Salem.

Lierman, 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.

Ligon, F. K., Dietrich, W.E. & Trush, W.J. (1995) Downstream ecological effects of dams: a geomorphic perspective. *BioScience* **45,**183–192.

Lilja, J. & Romakkaniemi, A. (2003) Early-season river entry of adult Atlantic salmon its dependency on environmental factors. *Journal of Fish Biology* **62,** 41–50.

Lindmark, E & Gustavsson, L.H. (2008) Field study of an attraction channel as entrance to fishways. *River Research & Applications* **24**, 564-570.

Linlokken, A. (1993) Efficiency of fishways and impact of dams on the migration of grayling and brown trout in the Glomma river system, South-eastern Norway. *Regulated Rivers: Research & Management* **8,** 145-153.

Lisle, T.E. (1989) Sediment transport and resulting deposition in spawning gravels, North Coastal California. *Water Resources Research,* **25,** 1303-1319.

Lobón-Cerviá, J. & Mortensen, E. (2005) Population size in stream-living juveniles of lake-migratory brown trout *Salmo trutta* L.: the importance of stream discharge and temperature. *Ecology of Freshwater Fish* **14**, 394-401.

Lohani, B., Evans, J.W., Ludwig, H., Everitt, R.R., Carpenter,R.A. & Tu, S.L. (1997) Environmental Impact Assessment for Developing Countries in Asia. Volume 1 - Overview. 356 pp.

Louhi, P., Maki-Petays, A. & Erkinaro, J. (2008) Spawning habitat of Atlantic salmon and brown trout: Generic criteria and intagravel factors. *River Research & Applications.* **24,** 330-339.

Lucas, M.C. & Baras, E. (2001) *Migration of Freshwater Fishes.* Oxford: Blackwell Science.

Lucas, M.C. & Frear, P.A. (1997) Effects of a flow-gauging weir on the migratory behaviour of adult barbel, a riverine, cyprinid. *Journal of Fish Biology* **50,** 382-396.

Lucas M.C., Mercer T., Batley E., Frear P.A., Peirson G., Duncan A. & Kubecka J. (1998) Spatio-temporal variations of fishes in the Yorkshire Ouse system. *Sci. Total Environ*. **210–211**, 437–455.

Lucas, M.C., Bubb, D.H., Jang, M.H., Ha, K. & Masters, J.E.G. (2009) Availability of and access to critical habitats in regulated rivers: effects of low-head barriers on threatened lampreys. *Freshwater Biology* **54**, 621-634.

Lundqvist, H., Rivinoja, P., Leonardsson, K. & McKinnell, S. (2008) Upstream passage problems for wild Atlantic salmon (*Salmo salar* L.) in a regulated river and its effect on the population. *Hydrobiologia* **602**, 111-127.

Lytle, D.A. & Poff, N.L. (2004) Adaptation to natural flow regimes. *Trends in Ecology and Evolution* **19,** 94–100.

Macleod, M., Moran, D., & Spencer, I. (2006). Counting the cost of water use in hydroelectric generation in Scotland. *Energy Policy*, **34**(15), 2048–2059.

Mader, H. & Maier, C. (2008) A method for prioritizing the reestablishment of river continuity in Austria. *Hydrobiologia* **609**, 277-288.

Maes, J., Turnpenny, A.W.H., Lambert, D.R., Nedwell, J.R., Parmentier, A. & Ollevier, F. (2004) Field evaluation of a sound system to reduce estuarine fish intake rates at a power plant cooling water inlet. *Journal of Fish Biology* **64**, 938-946.

Magoulick, D.D. (2000) Spatial and temporal variation in fish assemblages of drying stream pools: the role of abiotic and biotic factors. *Aquatic Ecology* **34**, 29–41.

Magoulick, D.D. & Kobza, R.M. (2003) The role of refugia for fishes during drought: a review and synthesis. *Freshwater Biology* **48**, 1186–1198.

Mainstone, C.P., Barnard, S. & Wyatt, R. (1994) Development of a Fisheries Classification Scheme. Report to NRA 244/7/Y. National Rivers Authority.

Maitland, P.S. (2004) Keys to the freshwater fish of Britain and Ireland, with notes on their distribution and ecology. Scientific publication No. 62, Freshwater Biological Association: Ambleside.

Mallen-Cooper, M. (1999) *Developing fishways for nonsalmonid fishes: a case study from the Murray River in Australia. In: M.Odeh (ed.) Innovations in Fish Passage Technology. Bethesda, MD: American Fisheries Society,* 173–195*.*

Mallen-Cooper, M. (2000) Taking the mystery out of migration. Proceedings of the 1990 annual workshop of the Australian society for Fish Biology, Bendigo, Victoria, 28-29 September 1999, 110-111.

March, J.G., Benstead, J.P., Pringle, C.M. & Scatena, F.N. (2003) Damming tropical island streams: Problems, solutions, and alternatives. *BioScience* **53,** 1069-1078.

Marchetti, M.P. & Moyle, P.B. (2001) Effects of flow regime on fish assemblages in a regulated California stream. *Ecological Applications* **11,** 530-539.

Masters, J.E.G., Jang, M.H., Ha, K., Bird, P.D., Frear, P.A. & Lucas, M.C. (2006) The commercial exploitation of a protected anadromous species, the river lamprey (*Lampetra fluviatilis* (L.)), in the tidal River Ouse, north-east England. *Aquatic Conservation: Marine and Freshwater Ecosystems* **16**, 77–92.

Matthews, W.J. (1985) Critical current speeds and microhabitats of the benthic fishes, Percina roanoka and Etheostoma flabellare. *Environmental Biology of Fishes* **12,** 303–308.

Matthews, W.J. & Marsh-Matthews, E. (2003) Effects of drought on fish across axes of space, time and ecological complexity. *Freshwater Biology* **48**, 1232–1253.

Mathers, R.G., De Carlos, M., Crowley, K. & Teangana, D. (2002) A review of the potential effect of Irish hydroelectric installations on Atlantic salmon (*Salmo salar*) populations, with particular reference to the River Erne. *Biology and the Environment: Proceedings of the Royal Irish Academy* **102B**, 69–79.

McKenzie, D.J.T. (2007) A layman's guide to small hydro schemes in Scotland. http://www.hi-energy.org.uk/documents/Laymans%20Guide%20to%20Hydro.pdf.

Meehan, W.R. (ed.), (1991) *Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats*. American Fisheries Society, Bethesda, Maryland.

Meldgaard, T., Nielson, E.E. & Loeschcke, V. (2003) Fragmentation by weirs in a riverine system: A study of genetic variation in time and space among populations of European grayling (*Thymallus thymallus*) in a Danish river system. *Conservation Genetics* **4**, 735-747.

Merkx, J.C.A. & Vriese, F.T. (2007) *Archimedean screw and fish passage study, Hooidonkse Mill.* VisAdvies Report VA2006_41, 28 pp.

Merrett, S. (2007) The Thames catchment: a river basin at the tipping point. *Water Policy* **9**, 393–404.

Millenium Ecosystem Assessment (2005) *Ecosystems and Human Well-Being.* Washington, DC.: Island Press. Available at: http://www.millenniumassessment.org/.

Mills, D.H. (1989) *Ecology and Management of Atlantic Salmon*. London: Chapman and Hall.

Minns, C.K. (1997) Quantifying 'no net loss' of productivity of fish habitats. *Canadian Journal of Fisheries and Aquatic Sciences* **54,** 2463–2473.

Minns, C.K., Kelso, J.R.M., Randall, R.G. (1996) Detecting the response of fish to habitat alterations in freshwater ecosystems. Canadian Journal of Fisheries and Aquatic Sciences, **53,** 403-414.

Mitchell, S.C. & Cunjack, R.A. (2007) Relationship of upstream migrating adult Atlantic salmon (salmo salar) and stream discharge within Catamaran brook, New Brunswick. *Canadian Journal of Fisheries & Aquatic Science* **64,** 563-573.

Moir, H.J., Soulsby, C. & Youngson, A. (1998) Hydraulic and sedimentary characteristics of habitat utilized by Atlantic salmon for spawning in the Girnock Burn, Scotland. *Fish. Man.Ecol*. **5,** 241–254.

Morita, K. & Yamamoto, S. (2002) Effects of habitat fragmentation by damming on the persistence of stream-dwelling charr populations. *Conservation Biology* **16,** 1318-1323.

Moursund, R.A., Bleich, M.D., Ham, K.D. & Mueller, R.P. (2003) Evaluation of the effects of extended length submerged bar screens on migrating juvenile Pacific lamprey (*Lampetra tridentata*) at John Day Dam in 2002. Report for Contract DE-AC06-76RL01830. Prepared by Pacific Northwest National Laboratory for U.S. Army Corps of Engineers. Portland, Oregon.

Murchie, K.J., Hair, K.P.E., Pullen, C.E., Redpath, T.D., Stephens, H.R. & Cooke, S.J. (2008) Fish response to modified flow regimes in regulated rivers: Research methods, effects and opportunities. *River Research and Applications* **24**, 197-217.

Nelson, V. (2011) Introduction to renewable energy. Energy and the Environment. Abbas Ghasserhi Series editor. Taylor and Francis Group, LLC.

Netboy, A. (1968) *The Atlantic Salmon, a Vanishing Species*? London: Faber and Faber.

Niemela¨, E., Erkinaro, J., Julkunen, M. & Hassinen, E. (2005) Is juvenile salmon abundance related to subsequent and preceding catches? Perspectives from a long-term monitoring programme. *ICES Journal of Marine Science* **62,** 1617-1629.

Nilsson, C. (1996) Remediating river margin vegetation along fragmented and regulated rivers in the north: what is possible? *Regulated Rivers: Research & Management* **12**, 415-431.

Nilsson, C & Brittain, J.E. (1996) Remedial strategies in regulated rivers: Introductory remarks. *Regulated Rivers: Research & Management* **12**, 347-351.

Nilsson, C., Reidy, C., Dynesius, M. & Revenga, C. (2005) Fragmentation and flow regulation of the world's large river systems. *Science* **308**, 405-408.

Noble, R.A.A. & Cowx, I.G. (2007) Development of a predictive fish community typology and sensitivity to abstraction rating system for England and Wales to fit into the RAM Framework review. Report to ENTEC and EA.

Noble, R.A.A., Bredeweg, B., Linnebank, F., Salles, P. & Cowx, I.G. (2009) A qualitative model of limiting factors for a salmon life cycle in the context of river rehabilitation. *Ecological Informatics* **4**, 299-319

Normandeau Associates, Inc., Skalski, J.R. & Mid Columbia Consulting, Inc. (1996) Fish survival investigation relative to turbine rehabilitation at Wanapum Dam, Columbia River, Washington. Report prepared for Grant County Public Utility District No. 2, Ephrata, Washington.

North, E. (1979) aggressive-behaviour of juvenile brown trout *Salmo-trutta*,L - analysis of the wigwag display. *Journal of Fish Biology* **15,** 571-577.

Northcote, T. G. (1984) Mechanisms of fish migration in rivers. In Mc Cleave, J.D., J.J. Dodson & W. H. Neill (eds) *Mechanisms of Migration in Fishes*. New York: Plenum, pp. 317–355.

Northcote, T.G. (1995) Comparative biology and management of Arctic and European grayling (*Salmonidae, Thymallus*) *Reviews in Fish Biology and Fisheries* **5**, 141–194.

Nunn, A.D., Harvey, J.P., Britton, J.R., Frear, P.A. & Cowx, I.G. (2007) Fish, climate and the Gulf Stream: the influence of abiotic factors on the recruitment success of cyprinid fishes in three English lowland rivers. *Freshwater Biology* **52**, 1576-1586.

Nunn, A.D., Harvey, J.P., Noble, R.A.A. & Cowx, I.G. (2008) Condition assessment of lamprey populations in the Yorkshire Ouse catchment, north-east England, and the potential influence of physical migration barriers. *Aquatic Conservation: Marine and Freshwater Ecosystems* **18**, 175-189.

Nunn, A.D., Copp, G.H., Vilizzi, L. & Carter, M.G. (2010) Seasonal and diel patterns in the migrations of fishes between a river and a floodplain tributary. *Ecology of Freshwater Fish* **19**, 153-162.

NWE. New World Encyclopaedia. (2011) http://www.newworldencyclopedia.org/entry/Turbine.

O'Connor, J.P., O'Mahony, D.J.& O'Mahoney, J.M.. (2005). Movements of *Macquaria ambigua,* in the Murray River, south-eastern Australia. *Journal Fish Biology.* **66,** 392-403.

O'Connor, J.P., O'Mahony, J.M. & Glenane, T.J. (2006) Some impacts of low and medium head weirs on downstream fish movement in the Murray-Darling basin in southeastern Australia. *Ecology of Freshwater Fish* **15**, 419-427.

Odeh, M. (1999a). Fish passage innovation for ecosystem and fishery restoration. In: Odeh, M., ed. Innovations in fish passage technology. Bethesda, Maryland: American Fisheries Society, 1-24.

Odeh, M. (1999b). A summary of environmentally friendly turbine design concepts. US Department of Energy Idaho Operations Office.

O'Hanley, J.R. & Tomberlin, D. (2005) Optimizing the removal of small fish passage barriers. *Env Model Assess* **10,** 85-98.

O'Keeffe, N. & Turnpenny, A.W.H. (2005) *Screening for intake and outfalls: a best practice guide.* Environment Agency Science Report SC030231, 154 pp.

Økland, F., Erkinaro, J., Moen, K., Niemelae, E., Fiske, P., McKinley, R.S. & Thorstad, E.B. (2001) Return migration of Atlantic salmon in the river Tana: phases of migratory behaviour. *Journal of Fish Biology* **59,** 862–874.

Old, G & Acreman, M. (2006) Guidance on compensation flows and freshets. Task 3:Literature review. Project WFD82 Centre for Ecology and Hydrology, Wallingford, UK. Edinburgh: SNIFFER.

Ormerod, S.J. (2003) Current issues with fish and fisheries: editor's overview and introduction. *Journal Applied Ecology* **40,** 204–213.

OSPAR 2010. Background document for European eel *Anguilla anguilla.* OSPAR Commission, London. Biodiversity Series, 29pp

Ovidio, M & Philippart, J.C. (2002) The impact of small physical obstacles on upstream movements of six species of fish. *Hydrobiologia* **483**, 55-69.

Paish, O. (2002) Small hydro power: technology and current status. Renewable & Sustainable *Energy Reviews* **6**, 537-556.

Paish, O. (2011). GPG presentation – Comment from the BHA. Conference outputs – Hydropower in England and Wales – Addressing Key Ecological Issues. 03/03/2011. SOAS, London.

Paquet, P.J & Witmer, G.W. (1985) Cumulative impacts of small hydropower developments: an overview of the issues. In: *Symposium on Small Hydropower and Fisheries.* Denver., Colorado. pp. 343-345.

Pavlov, D.S. (1989) Structures assisting the migration of non-salmonid species. USSR. FAO fisheries Technical paper No.308. Rome.
Pavlov, D.S., Lupandin, A.I. & Kostin, V.V. (2002) downstream migration of fish through dams of hydroelectric power plants. Oak Ridge National Laboratory: Oak Ridge.
Penczak, T. & Kruk, A. (2000) Threatened obligatory riverine fishes in human-modified Polish rivers. *Ecology of Freshwater Fish* **9**, 109–117.
Peter, A. (1998) Interruption of the river continuum by barriers and the consequences for migratory fish. In: M. Jungwirth, S. Schmutz. & S. Weiss (eds) *Fish Migration and Fish Bypasses*. Oxford: Fishing News Books, Blackwell Publications, pp. 99–112.
Petts, G.E. (1984) *Impounded Rivers.* Chichester: John Wiley & Sons.
Petts, G.E. (1990) Regulation of large rivers: problems and possibilities for environmentally sound river development in South America. *Interciencia* **15,** 388–395.
Petts, G. E. (1996) Water allocation to protect river ecosystems. *Regulated Rivers: Research and Management* **12,** 353-365.
Pielou, E.C. (1998) *Freshwater.* Chicago: The University of Chicago Press.
Poff, N.L. & Hart, D.D. (2002) How dams vary and why it matters for the emerging science of dam removal. *BioScience* **52**, 659-668.
Poff, N.L. & Zimmerman, J.K.H. (2010) Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. *Freshwater Biology* **55**, 194–205.
Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard K.L., Richter, B.D., Sparks, R.E. & Stromberg, J.C. (1997) The natural flow regime: a paradigm for river conservation and restoration. *BioScience* **47**, 769-784.
Poff, N.L., Olden, J.D., Merritt, D.M. & Pepin, D.M. (2007) Homogenization of regional river dynamics by dams and global biodiversity implications. Proceedings of the National Academy of Sciences of the United States of America, **104**, 5732–5737.
Postel, S. (2005) Liquid Assets: the Critical Need to Safeguard Freshwater Ecosystems. Worldwatch Paper 170, Worldwatch Institute, Washington, DC.
Postel, S. & Carpenter, S. (1997) Freshwater ecosystem services. In: G.C. Daily (ed.) *Nature's Services: Societal Dependence on Ecosystems*. Washington, DC: Island Press, pp. 195–214.
Postel, S. & Richter, B. (2003) *Rivers for Life: Managing Water for People and Nature*. Washington, DC: Island Press.
Potter, E.C.E. (1988) Movements of atlantic salmon salmo salar L., in an estuary in south west England. *Journal of Fish Biology* **33,** 153-159
Poulet, N. (2007) Impact of weirs on fish communities in a piedmont stream. *River Research & Applications* **23**, 1038-1047.
Power M.E., Sun, A., Parker, G., Dietrich, W.E. & Wootton J.T. (1995) Hydraulic food-chain models. *BioScience* **45**, 159–167.
Power, M.E., Dietrich, W.E & Finlay, J.C. (1996) Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. *Environmental Management* **20,** 887–895.
Pre´vost, E. & Chaput, G., eds. (2001) Stock, recruitment and reference points: assessment and management of Atlantic salmon. Paris: Hydrobiology and Aquaculture, INRA Editions. 223 pp.
Pringle, C.M. (1997) Exploring how disturbance is transmitted upstream: going against the flow. *Journal of the North American Benthological Society* **16,** 425–438.
Prosek, J. 2010. A steady, steep decline for the lowly, uncharismatic Eel. Yale environment 360. Opinion, Analysis, reporting & debate. Yale school of forestry & environmental studies.
Pusey, B.J., Arthington, A.H. & Read, M.G. (1993) Spatial and temporal variation in fish assemblage structure in the Mary River, south-east Queensland: the influence of habitat structure. *Environmental Biology of Fishes* **37**, 355–380.
Radford, A., Riddington, G. & Anderson, J. (2004) *Research Report: The Economic Impact of Game and Coarse Angling in Scotland* [Internet], Edinburgh, The Scottish Government Publications. Available: http://www.scotland.gov.uk/Publications/2004/03/19079/34369
Ramos, H. & de Almeida, A.B. (1999) Small hydropower schemes as an important renewable energy source. In: Proceedings of the Hydroenergia 99, Vienna, Austria, 11–13 October.
Reeves, G.H., Benda, K.M., Burnett, P.A. & Sedell, J.R. (1995) A disturbance based approach to maintaining and restoring freshwater habitats of evolutionary significant units of anadromous salmonids in the Pacific Northwest. pp 334-349 in Nielson, J.L. Evolution

and the aquatic system: defining unique units in population conservation. America fisheries Society, Symposium, 17, Bethesda, Maryland.
REFOCUS (2002) Working Together: Understanding the effects of small hydropower. Elsevier. Volume 2002, Number 7, July 2002, pp. 58-59(2).
Renofalt, B.M., Jansson, R. & Nilsson, C. (2010) Effects of hydropower generation and opportunities for environmental flow management in Swedish riverine ecosystems. *Freshwater Biology* **55**, 49-67.
Richkus, W.A. & Dixon, D.A. (2003). Review of research and technologies on passage and protection of downstream migrating catadromous eels at hydroelectric facilities. D.A. Dixon (ed.) *Biology, Management and Protection of Catadromous Eels.* Symposium 33. Bethesda, MD, USA: American Fisheries Society, pp. 377-388.
Richter, B.D., Baumgartner, J.W., Wigington, R. & Braun, D.P. (1997) How much water does a river need? *Freshwater Biology* **37,** 231-249.
Richter, B.D., Matthews, R., Harrison, D.L. & Wigington, R. (2003) Ecologically sustainable water management: managing river flows for ecological integrity. *Ecological Applications* **13**, 206-224.
Righton, D., Aarestrup, K., Jellyman, D., Sebert, P., Van Den Thillart, G. & Tsukamoto, K. (2012) The *Anguilla* spp. Migration problem: 40 million years of evolution and two millennia of speculation. *Journal Fish Biology* **81,** 365-386.
Rivinoja P., McKinnell, S. & Lundqvist, H. (2001) Hindrances to upstream migration of Atlantic salmon in a Northern Swedish river caused by a hydroelectric power-station. *Regulated Rivers: Research and Management* **17,** 101-115.
Rizzo, B. (1985) Fish migration at small hydropower developments: an overview of the issues. In: *Symposium on Small Hydropower and Fisheries.* Denver., Colorado, May 1985. pp. 241-242.
Robertson, M.J., Pennell, C.J., Scruton, D.A., Robertson, G.J. & Brown, J.A. (2004) Effect of increased flow on the behaviour of Atlantic salmon parr in winter. *Journal of Fish Biology* **65**, 1070-1079.
Robson A., Cowx I.G. & Harvey J.P. (2012) Impact of run-of-river hydro-schemes upon fish populations. Phase 2 High-head hydropower scheme sampling. SNIFFER WFD114 project report 223 pp.
Roni, P. (2005) Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.
Roni P., Liermann M. & Steel, A. (2003) Monitoring and evaluating responses of salmonids and other fishes to instream restoration. Pages 318-329 in Montgomery D.R., Bolton S. and Booth D.B, editors. Restoration of Puget Sound rivers. University of Washington Press, Seattle.
Rosenberg, D.M., Bodaly, R.A. & Usher, P.J. (1995) Environmental and social impacts of large-scale hydroelectric development: who is listening? *Global Environmental Change* **5,** 127–148.
Rosenberg, D.M., Berkes, F., Bodaly, R.A., Hecky, R.E., Kelly, C.A. & Rudd, J.W.M. (1997) Large-scale impacts of hydroelectric development. *Environmental Reviews* **5,** 27–54.
Sabaton, C., Souchon, Y., Capra, H., Gouraud, V., Lascaux, J.M. & Tissot, L. (2008) Long-term brown trout populations responses to flow manipulation. *River Research & Applications* **24**, 476-505.
Sala, O.E., Meyerson, L.A. & Parmesan, C. (eds) (2008) *Biodiversity Change and Human Health.* Washington, DC: Island Press.
Saltveit, S.J., Halleraker, J.H., Arnekleiv, J.V. & Harby, A. (2001) Field experiments on stranding in juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) during rapid flow decreases caused by hydropeaking. *Regulated Rivers: Research & Management* **17**, 609–622.
Sambrook, H.T. & Cowx, I.G. (2000) Wimbleball pumped storage scheme: integration of water resource management, engineering design and operational control to compliment the needs of the salmonid fisheries of the River Exe, UK. In: I.G. Cowx (ed.) *Management and Ecology of River Fisheries* Oxford: Fishing News Books, Blackwell Science, pp 177-200.
Saunders, J.W. (1960) The effect of impoundment on the population and movement of atlantic salmon in the Ellerslie Brook, Prince Edward Island. *Journal Fish Research Board Can* **17,** 453-473.
Saunders, R.L. & Schom, C.B. (1985) Importance of the variations in life history parameters of Atlantic Salmon (*Salmo salar). Canadian Journal of Fisheries and Aquatic Sciences*. **42,** 615-618.

Schaffer, W. M. (1974) Selection for optimal life histories: the effects of age structure. *Ecology* **55**, 291–303.
Schilt, C.R. (2007) Developing fish passage and protection at hydropower dams. *Applied Animal Behaviour Science* **104**, 295-325.
Schlosser, I. J. (1982) Fish community structure re and function along two habitat gradients in a headwater stream. *Ecological Monographs* **52,** 395–414.
Schmidt, J. (1922) The breeding places of the eel. *Philosophical Transactions of the Royal Society of London* Series B, **211**, 179–208.
Schmutz, S, Cowx, I.G., Haidvogl, G. & Pont, D. (2007) Fish-based methods for assessing European running waters: a synthesis. *Fisheries Management and Ecology* **14**, 369-380.
Scott, W.B. & Crossman, E.J. (1998). Freshwater fishes of Canada, 4th edn. Oakville, ON: Galt House Publications Ltd.
Scottish Office (1997) Report of the Scottish Salmon Strategy Task Force. Scottish Salmon Strategy Task Force. Edinburgh: Scotland.
Sear, D.A., Frostick, L.B., Rollinson, G. & Lisle, T.E. (2008) The significance and mechanics of fine sediment infiltration and accumulation in gravel spawning beds. *American Fisheries Society Symposium,* **65,** 149-173.
Sedgwick, R.W. (2006) Manual of Best Practice for Fisheries Impact Assessments. Environment Agency Report. Science Report SC020025/SR. 129pp.
Semple, J.R. (1991) Atlantic salmon habitat survey: enhancement opportunities and problems in the Dunbar stream, Naswaak river, New Brunswick. Can. Manuscript. Rep. Fish. Aquat. Sci. No. 2076, pp. 1–35
SEPA (2008) The Water Environment (Controlled Activities) (Scotland) Regulations 2005. A Practical Guide, 31 pp. CAR Practical Guide 2008 http://www.sepa.org.uk/water/idoc.ashx?docid=2a4b7ea0-ccf7-48b5-a85b 1404de0f58ea&version=-1
SEPA (2010a) Guidance for Developers of Run-of-River Hydropower Schemes. SEPA, Edinburgh, Scotland. 26pp.
SEPA (2010b) SEPA's consultation on its Guidance for Developers of Run-of-River Hydropower Schemes. Response to consultees. SEPA, Edinburgh, Scotland. 8pp.
SEPA (2011) http://www.sepa.org.uk/default.aspx. Accessed 2011.
Shiao, J.C., lozys, L., iizuka, Y. & Tzeng, W.N. (2006) Migratory patterns and contribution of stocking to the population of European eel in Lithuanian waters as indicated by otolith Sr:Ca ratios. Journal of *Fish Biology* **69,** 749-769.
Smith, G.W., Smith, I.P., Armstrong, S.M (1994) The relationship between river flow and entry to the Aberdeenshire Dee by returning adult Atlantic salmon. *Journal fish biology* **50** 463-474.
Smith, P.E., Orvos, D.R. & Cairns, J. (1993) Impact assessment using the before-after-control-impact (BACI) model: concerns, and comments. *Canadian Journal of Fisheries and Aquatic Sciences* **50,** 627-637.
SNH (2012) http://www.snh.gov.uk/land-and-sea/managing-freshwater/freshwater-fisheries/.
Solomon, D.J. (1992) Diversion and entrapment of fish at water intakes and outfalls. National Rivers Authority, Bristol. R & D Report 1.
Solomon, D.J., Sambrook, H.T. & Broad, K.J. (1999). *Salmon migration and river flow*. RD Publication 4, Environment Agency, Bristol.
Spah, H. (2001) Fishery biological opinion of the fish compatibility of the patented hydraulic screw from Ritz Atro. Bielfield, Germany.
Sprengel, G. & Lüchtenberg, H. (1991) Infection by endoparasites reduces maximum swimming speed of European smelt *Osmerus eperlanus* and European eel A*nguillaanguilla*. *Diseases of Aquatic Organisms* **11,** 31-35.
Stanford, J.A., Ward, J.V., Liss, W.J., Frissell, C.A., Williams, R.N., Lichatowhich, J.A. & Coutant, C.C. (1996) A general protocol for restoration of regulated rivers. *Regulated Rivers: Research & Management* **12**, 391–413.
Stenseth, N.C. & Mysterud, A. (2002) Climate, changing phenology and other life history traits: Nonlinearity and match-mismatch to the environment. Department of Biology, Division of Zoology, University of Oslo, pp. 13379-13381.
Stradmeyer, L., Hojesjo, J., Griffiths, S.W., Gilvear, D.J. & Armstrong, J.D. (2008) Competition between brown trout and Atlantic salmon parr over pool refuges during rapid dewatering. Journal of *Fish Biology* **72,** 848-860.
Stuart, I.G. & Berghuis, A.P. (2002) Upstream passage of fish through a vertical-slot fishway in an Australian subtropical river. *Fisheries Management & Ecology* **9**, 111-122.

Tennant D.L. (1976) Instream flow regimens for fish, wildlife, recreation, and related environmental resources. In: J.F. Orsborn and C.H. Allman (eds) *Proceedings of the Symposium and Speciality Conference on Instream Flow Needs II.* Bethesda, Maryland: American Fisheries Society, pp. 359-373.

Tesch, F.W. (2003) *The Eel*, 3rd edn. Blackwell Publishing, Oxford, U.K.

Tetzlaff, D., Soulsby, C., Youngson, A.F., Gibbins, C., Bacon.P.J., Malcolm, I.A. & Langan, S. (2005) Variability in stream discharge and temperature: a preliminary assessment of the implications for juvenile and spawning Atlantic salmon. *Hydrol. Earth System. Science* **91,** 193-208.

The Water Environment (Controlled Activities) (Scotland) Regulations 2011 (CAR) Guidance for applicants on supporting information requirements for hydropower applications. 74pp. Available at: http://www.legislation.gov.uk/ssi/2011/209/pdfs/ssi_20110209_en.pdf.

Therrien, J & Bourgeois, G. (2000) Fish Passage at Small Hydro Sites. Report by Genivar Consulting Group for CANMET Energy Technology Centre, Ottawa, 114 p.

The Tay Western Catchments Project. (2010). A journey along 1000 miles of rivers and watercourses in Glen Lyon, Glen Lochay and Glen Dochart. Western Catchments. Big Sky Print Ltd. 50pp.

Thorstad, E.B., Okland, F., Kroglund, F. & Jepsen, N. (2003) Upstream migration of Atlantic salmon at a power station on the River Nidelva, Southern Norway. *Fisheries Management & Ecology* **10**, 139-146.

Trépanier, S., Rodriguez, M.A.& Magnan, P. (1996) Spawning migrations in landlocked Atlantic salmon: time series modelling of river discharge and water temperature effects. *Journal of Fish Biology* **48,** 925–936.

Trussart, S., Messier, D., Roquet, V. & Aki, S. (2002) Hydropower projects: a review of most effective mitigation measures. *Energy Policy* **30**, 1251-1259.

Tsukamoto, K. (2009). Oceanic migration and spawning of anguillid eels. *Journal Fish Biology* ***74,*** 1833-1852.

Turnpenny, A.W.H. (1989) Impoundment and abstraction – exclusion of fish from intakes. In: J. Gregory (ed.) *Water Schemes: the Safeguarding of Fisheries.* Pitlochry Atlantic Salmon Trust.

Turnpenny, A.W.H., Struthers, G. & Hanson, K.P. (1998) A UK guide to intake fish-screening regulations, policy and best practice. With particular reference to hydroelectric power schemes. With particular reference to hydroelectric power schemes. Fawley Aquatic Research Laboratories LTD & Hydroplan.

Ugedal, O., Naseje, T.F., Thorstad, E.B., Forseth, T., Saksgard, L.M. & Heggberget, T.G. (2006) Twenty years of hydropower regulation in the River Alta: long-term changes in abundance of juvenile and adult Atlantic salmon. *Hydrobiologia* **609**, 9-23.

Valentin, S., Lauters, F., Sabaton, C., Breil, P. & Souchon, Y. (1996) Modelling temporal variations of physical habitat for brown trout (*Salmo trutta*) in hydropeaking conditions. *Regulated Rivers: Research & Management* **12**, 317-330.

Van den berghe, E.P.& Gross, M.R. (1989) Natural selection resulting from female breeding competition in a pacific salmon (coho: *Oncorhynchus kisutch*) *Evolution* **43** 125-140.

Vehanen, T., Bjerke, P.L., Heggenes, J., Huusko, A. & Maki-Petays, A. (2000) Effect of fluctuating flow and temperature on cover type selection and behaviour by juvenile brown trout in artificial flumes. *Journal of Fish Biology* **56**, 923-937.

Verspoor, E., Stradmeyer, L. & Nielsen, J.L. (2007) T*he Atlantic Salmon. Genetics, Conservation and Management.* Oxford: Blackwell Publishing, 280 pp.

Vovk-Korz. A., Korze, D., Goltara, A., Conte, G., Toniutti, N. & Smolar-Zvanut, N. (2008) Review of best practices/guidelines for compensation measures for hydropower generation facilities. Certification for hydro: Improving clean energy. Intelligent energy, Europe.

Walker, K.F. & Thoms, M.C. (1993) Environmental effects of flow regulation on the lower River Murray, Australia. *Regulated Rivers: Research & Management* **8**, 103-119.

Walker K.F., Sheldon F. & Puckridge J.T. (1995) A perspective on dryland river ecosystems. *Regulated Rivers: Research & Management* **11**, 85–104.

Ward, J.V. (1989) The four dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society* **8,** 2–8.

Ward, J.V. & Stanford, J.A. (1979) *The Ecology of Regulated Streams.* New York: Plenum Press.

Ward, J.V. & Stanford, J.A. (1995) Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. *Regulated rivers: Research & Management* **11**, 105-119.

Warnken, J. & Buckley, R. (1998) Scientific quality of tourism environmental impact assessment. *Journal of Applied Ecology* **35,** 1-8.

Watene, E.M., Boubee, J.A.T. & Haro A. (2003) Downstream movement of mature eels in a hydroelectric reservoir in New Zealand. In: D.A. Dixon (ed.) *Biology, Management and Protection of Catadromous Eels,* Symposium 33. Bethesda, MD, USA: American Fisheries Society, pp. 295–305.

Weaver, C. (1963) Influence of water velocity upon orientation and performance of adult migrating salmonids. *U.S. Fish and Wildlife Service Fishery Bulletin* **63,** 97–121.

Webb, J. (1990) The behaviour of adult Atlantic salmon ascending the Rivers Tay and Tummel to Pitlochry dam. *Scottish Fisheries Research Report* **48,** 1-27.

Welcomme, R.L., Bene, C., Brown, C.A., Arthington, A.H., Dugan, P., King, J.M. & Sugunan V. (2006) Predicting the water requirements of river fisheries. *Ecological Studies* **90,**123–154.

Whiting, P.J. (2002) Streamflow necessary for environmental maintenance. *Annual Review of Earth and Planetary Sciences* **30**,181–206.

Williams, D. (2010) The British hydropower association perspective. An environment agency – CIWEM conference. Hydropower opportunities, challenges and sharing best practice. June 23rd 2010. SOAS, London.

Winston, M.R., Taylor, C.M. & Pigg, J. (1991) Upstream extirpation of four minnow species due to damming of a prairie stream. *Transactions of the American Fisheries Society* **120,** 98–105.

Winter, H.V., Jansen, H.M. & Bruijs, M.C.M. (2006) Assessing the impact of hydropower and fisheries on downstream migrating silver eel, *Anguilla anguilla,* by telemetry in the River Meuse. *Ecology of Freshwater Fish* **15,** 221-228.

Wootton, J.T, Parker, M.S. & Power, M.E. (1996) Effects of disturbance on river food webs. *Science* **273,** 1558–1561.

Wyatt R.J., Barnard S. & Lacey R.F. (1995) Use of HABSCORE V software and application to impact assessment. Report to NRA, **No. 400**; WRc.

Youngson, A.F. & Hay, D.W. (1996) The Lives of Salmon. Swan-Hill Press, Shrewsbury.

Zippin C. (1956) An evaluation of the removal method for estimating animal populations. *Biometrics* **8**, 163-189.

.